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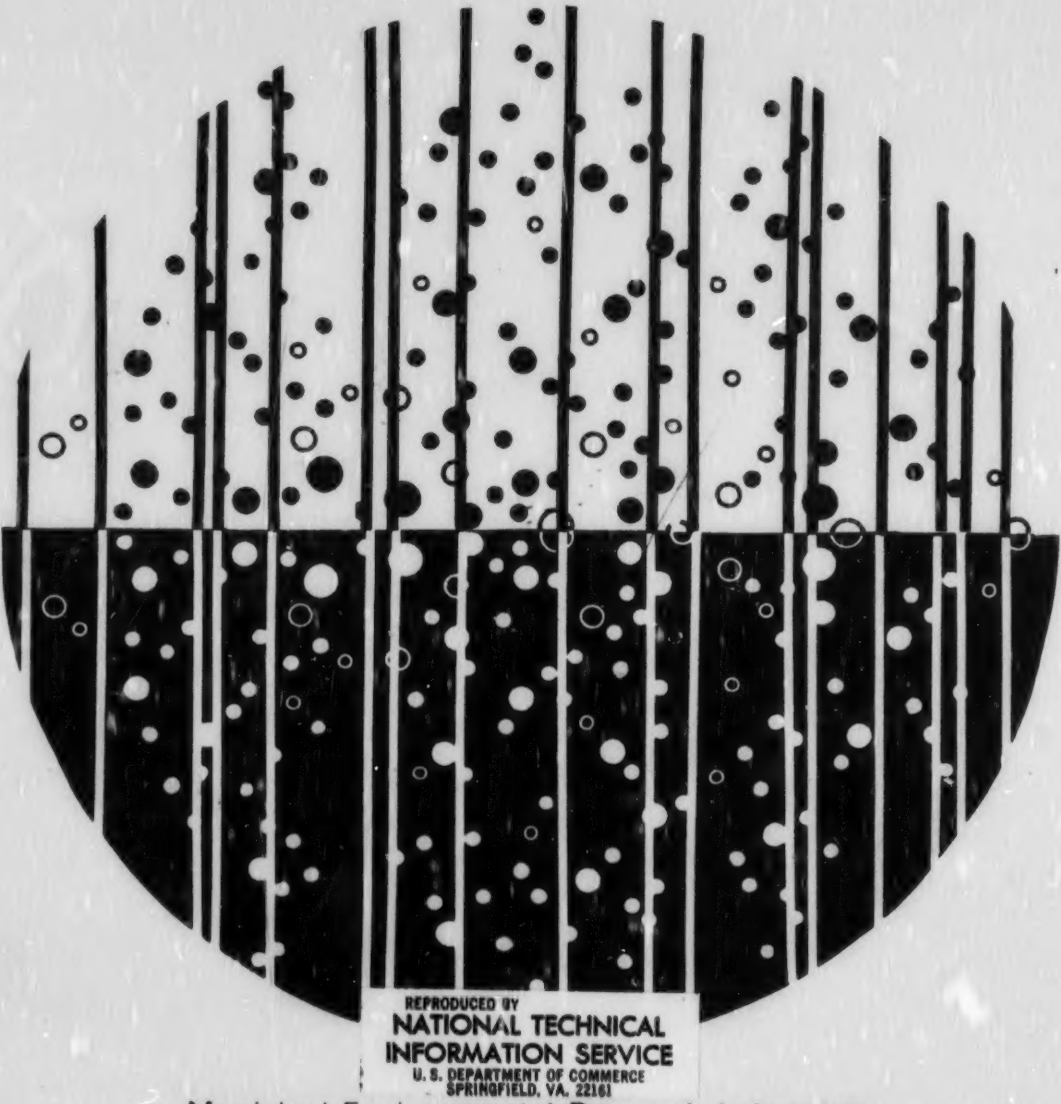
Proceedings of the Third Annual Municipal Solid Waste Research Symposium



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16. ABSTRACT

A research symposium on the mangement of gas and leachate produced from sanitary landfills was held to disseminate latest research informations on the subject, and exchange ideas between symposium participants from various parts of the U.S. and Canada. Topics under discussion included gas and leachate formation, collection, management and treatment. The papers contained in this symposium presents the Solid and Hazardous Waste Research Division, Municipal Environmental Research Laboratory, research on sanitary landfills. Selected papers from other organizations were included to identify closely related work.

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Department of Civil Engineering, University of Missouri--Columbia

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FOREWORD

The Environmental Protection Agency was created because of increasing public and government concern about the dangers of pollution to the health and welfare of the American people. Noxious air, foul water, and spoiled land are tragic testimony to the deterioration of our natural environment. The complexity of the environment and the interplay between its components require a concentrated and integrated attack on the problem.

Research and development is the necessary first step in problem solution and it involves defining the problem, measuring its impact, and searching for solutions. The Municipal Environmental Research Laboratory develops new and improved technology and systems for the prevention, treatment, and management of wastewater and solid and hazardous waste pollutant discharges from municipal and community sources, for the preservation and treatment of public drinking water supplies, and to minimize the adverse economic, social, health and aesthetic effects of pollution. This publication is one of the products of that research; a most vital communications link between the researcher and the user community.

The proceedings identifies research aimed at management of gas and leachate formed in sanitary landfills and provides solutions to these unique problems.

Francis T. Mayo
Director
Municipal Environmental
Research Laboratory

ABSTRACT

The symposium proceedings are primarily intended to disseminate up-to-date information on extramural research on gas and leachate formation, collection and management in sanitary landfills funded by the Solid and Hazardous Waste Research Division (SHWRD), U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory in Cincinnati, Ohio. Selected papers from work of other organizations were included in the symposium to identify closely related work not included in the SHWRD program.

The proceedings should be of value to researchers, designers, planners, and governmental agencies to select and manage landfill sites that will prevent environmental contamination from leachate and gas production.

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CURRENT RESEARCH ON LAND DISPOSAL OF MUNICIPAL SOLID WASTES

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ABSTRACT

The Solid and Hazardous Waste Research Division (SHWRD), Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, in Cincinnati, Ohio, has responsibility for research in the areas of municipal solid and hazardous waste management, including both disposal and processing. This research is being directed towards new and improved systems of municipal solid and hazardous waste management, development of technology, determination of environmental effects, and collection of data necessary for the establishment of processing and disposal guidelines.

Division activities in the area of municipal solid waste research have related to storage, collection, transport, processing, resource recovery, and disposal. Recent emphasis on energy has resulted in an expansion of the waste-as-fuels program.

The current municipal solid waste disposal research program has been divided into three general areas: (1) Pollutant Predictions for Current Landfill Techniques, (2) Alternatives to Current Landfill Disposal Techniques, and (3) Remedial Action for Minimizing Pollutants from Unacceptable Sites.

The research activites currently funded under these three general areas have been classified into seven categories shown below:

1. Residual Characterization/Decomposition
2. Pollutant Transport
3. Pollutant Control/Treatment
4. Co-disposal
5. Environmental Assessment
6. Remedial Action
7. Landfill Alternatives

INTRODUCTION

Increasing amounts of waste residuals are being directed to the land for disposal in landfills. At the same time, there is increasing evidence of environmental damage resulting from improper operation. The burden of operating landfills and coping with any resulting damages falls most heavily on municipalities and other local government agencies. Their problems are complex, involving legislation, economics, and public attitudes as well as technology; additionally, comprehensive information on landfilling techniques and protect ion of the local environment is not readily available.

Current estimates indicate that 144 million tons (as generated with moisture) of municipal wastes and 260 million tons (dry) of industrial wastes are disposed of to the land. A survey of national solid waste management practices conducted in 1968 by the Federal government through cognizent state agencies indicated less than 6 percent of 6000 land disposal sites surveyed could be considered sanitary landfills, based on very modest criteria not including a re-evaluation of groundwater pollution. More recent surveys (in 1975 and 1977) indicated a 25 percent increase over a 2-year period in the number of disposal sites considered to be sanitary landfills (5740 of 15,821 sites in 1976). The total number of disposal sites appears to have increased greatly since 1968, but this can be explained, in part by better record keeping by state agencies and/or that environmental enactments have both severely restricted refuse disposal by burning or dumping at sea and greatly increased the amount of solid waste generated as residues from air and water cleaning operations. The most pernicious effect of unsound disposal is the contamination of groundwater by leachate; about half of the United States domestic water supply is from groundwater. Groundwater contamination is usually discovered long after the damage is done and too late for corrective measures.

The municipal waste disposal program initiated by SHWRD was designed to document and evaluate the potential adverse environmental and public health effects that could result from application of waste disposal methods without proper precautions for leachate and gas management. The information thus obtained will provide the necessary data for the establishment of guidelines for communities to develop economical and environmentally safe municipal waste disposal management systems.

Specifically, in the area of landfill disposal techniques, a comprehensive data base on the characteristics of municipal and hazardous wastes will be developed to assess pollutants within a waste residual, pollutant release from waste residuals in the form of leachates and gas, decomposition rates under varying moisture regimes, the migration and attenuation of pollutants, and the resultant environmental damages. In addition, liner materials, both natural and synthetic, and chemical stabilization techniques for controlling leachate movement will be investigated. Research efforts will be conducted primarily through laboratory and pilot studies with some field testing of laboratory-based results.

In the area of alternative land disposal methods, technical and environmental data will be obtained to provide a basis for logical engineering decisions on viable environmentally sound methods other than landfill.

In the area of remedial action for preventing pollutant generation from unacceptable landfill sites, an engineering feasibility and design plan will be developed and tested in a field verification study.

This research strategy, encompassing state-of-the-art documents, laboratory analysis, bench and pilot studies, and full-scale field verification studies, is at various stages of implementation, but over the next 5 years the research reports developed will be compiled as criteria and guidance documents for user communities. Also, the waste disposal research program will develop and compile a research criteria data base for use in the development of guidelines and standards for waste residual disposal to the land as mandated by the recently enacted legislation entitled "Resource Conservation and Recovery Act of 1976" (RCRA).

RESIDUAL CHARACTERIZATION/DECOMPOSITION

Studies in this area involve collecting composition data on municipal and hazardous wastes from individual waste residuals and landfill disposal sites. Sampling techniques, analytical methods/procedures, and waste compatibility and waste decomposition information will be developed for implementing better disposal practices and waste management.

The objectives of this research activity are to (1) quantify the gas and leachate production from current best-practice, sanitary landfilling and (2) modify the landfill method to reduce the environmental impact of gas and leachate production in a positive and predictable manner. These objectives are to be achieved by construction and long-term monitoring of typical and simulated landfill cells and investigation, development, and optimization of those factors that control gas and leachate production. Results are expected only after long-term monitoring, due to the extremely slow reaction rates.

Several previous reviews of these efforts have been presented. See Schomaker, N.B. and Roulier, M. H., Current EPA Research Activities in Solid Waste Management: Research Symposium on Gas and Leachate from Landfills: Formation, Collection and Treatment, March 25-26, 1975, Rutgers, State University of New Jersey; and Schomaker, N. B., Current Research on Land Disposal of Hazardous Wastes: Residual Management by Land Disposal: Proceedings of the Hazardous Waste Research Symposium, February 2-4, 1976, University of Arizona.

Standard Analytical Techniques

Analysis of the contaminants within a waste leachate sample is difficult due to interfering agents. Existing instrumentation functions well in the analysis of simple mixtures at low concentrations but interference problems can be encountered for complex mixtures at high concentrations (1 percent by weight and greater). In this range the sample cannot always be analyzed directly and commonly must be diluted and/or analyzed by the method of standard additions. Options are the development of standard procedures for diluting and accounting for errors introduced thereby or the development of instrumentation capable of accurate, direct measurements at high concentrations in the presence of potential multiple interferences. Existing USEPA procedures for water and wastewaters are often not applicable. Analytical procedures are being developed on an as-needed basis as part of the SHWRD projects. However, most of this work is specific to the wastes being studied and separate efforts were required to insure that more general procedures/equipment would be developed. A compilation of analytical techniques [1]* used for contaminant analysis has been published in a report entitled Compilation of Methodology Used for Measuring Pollution Parameters of Sanitary Landfill Leachates, EPA-600/3-75-011, October 1975, and SHWRD is currently conducting a collaborative testing study[2] on leachate analyses. In this study, leachate samples will be sent to some 40 or 50 laboratories for analysis of specific parameters. The results will provide information on detection limits and precision for contaminants in leachate, using currently accepted methods developed for water and wastewater.

Standard Leaching Tests

In studying the potential environmental impact of contaminants, a standard leaching test is needed to assess contaminant release from a waste. Such a test must provide information on the initial release of contaminants from a waste contacted not only by water but also by other solvents that could be introduced in disposal. Additionally, such a test must provide some estimate of the behavior of the waste under extended leaching. Experience from ongoing SHWRD projects indicates that some wastes may initially release only small amounts of contaminants, but, under extended leaching, will release much higher concentrations. Such leaching behavior has an impact on disposal regulation and on management of a disposal site, and information on this behavior must be obtained and used in classifying a waste. The Office of Solid Waste (OSW) has funded the Industrial Environmental Research Laboratory (IERL), USEPA project[3] to examine this background area and develop procedures for determining whether a waste contains significant levels of toxic contaminants and whether a waste will release such contaminants under a variety of leaching conditions.

Validation of a Standard Leaching Test (SLT) is planned for future efforts, funded in part by SHWRD.[2] The passage of the RCRA on October 21, 1976, imposing time restraints, necessitated developing an Interim Standard Leaching Test (ISLT). Existing leaching tests will be evaluated for those elements that may be of special benefit to the development of an SLT, and at least three candidate ISLT's will be chosen for further testing as part of the OSW/IERL project.

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*Superscript numbers refer to the project officers, listed immediately following this paper, who can be contacted for additional information.

Waste Leachability

The characteristics of leachates from municipal refuse and mixtures of municipal refuse and selected industrial and municipal sludges are being studied in several different projects. Leaching study results[4] for 437 tons of municipal refuse are being compared to results of leaching from 117-ton and 3-ton experimental landfills located at the Boone County Field Site (BCFS) of USEPA. Another study,[4] performed under contract at the Center Hill Facility of USEPA, involves comparing the characteristics of leachates obtained from 3-ton experimental landfills containing municipal refuse and selected sludges of municipal and industrial origin.

Results obtained from the large experimental landfill at BCFS indicated greater removal of contaminants through leaching than previously reported from other experimental landfills. The test cell, receiving approximately 21 in/yr of net infiltration and constructed in June of 1971, still leaches appreciable quantities of COD (4000 mg/l), Fe (350 mg/l), total solids (4500 mg/l), and other contaminants. Additionally, CO_2 and CH_4 are still produced. This evaluation of current disposal practices is to be completed in FY 1980.

Leachates were also assayed for fecal coliform and fecal streptococci. There was an initial large number of such organisms (>106 organisms/ml) present in the leachates, but the fecal coliform rapidly (over several months) dropped to low (<20 organisms/ml) levels. Fecal streptococci have continued to be present (10^3 to 10^6 organisms/ml) in the leachates throughout the 5 years of operating data at BCFS.

Survival of poliovirus in landfilling refuse was investigated at BCFS and the Center Hill Facility. Studies by USEPA showed the presence of this virus in municipal solid waste and the presence of the virus in leachates when the waste was surcharged with large volumes of water. Poliovirus-seeded refuse samples were exposed to landfill conditions for 10 or more days and no poliovirus was recovered. The high ambient temperatures (air = 35°C; refuse = 59°C) were assumed to be the principal cause of virus inactivation. No viruses were found in the leachate. Leachate was, however, found to be antagonistic to seeded viruses. Temperature of the leachate (5°, 10°, 15°C) was found to be very important in determining the rate of inactivation. The survival of poliovirus within landfilling refuse was repeated under different ambient conditions (air = 0°C; refuse = 15°, 18°, and 27°C); survival was found to be temperature dependent and 33 days after the refuse was landfilled survival was less than 1 percent. The viral and bacterial efforts at BCFS have been reported in the News of Environmental Research in Cincinnati, "Survival of Fecal Coliforms and Fecal Streptococci in a Sanitary Landfill" (April 12, 1974) and "Poliovirus and Bacterial Indicators of Fecal Pollution in Landfill Leachates" (January 31, 1975).

Waste Decomposition

Waste decomposition data are being obtained from several ongoing efforts. One study[4] is a modification of the landfill method to accelerate waste decomposition in a predictable manner. This study was successfully performed on a laboratory scale and a report entitled Sanitary Landfill Stabilization with Leachate Recycle and Residual Treatment, EPA-600/2-75-043, October 1973, has been published. Leachate recirculation with pH control results in waste decomposition in a time period as short as 6 months. Field application will probably yield a 1- or 2-year decomposition period due to anticipated leachate distribution problems.

A second effort[4] involves the elucidation of the role of moisture regime (different net infiltration conditions). This study is being performed on a laboratory scale. It will yield valuable information with respect to the kinetics of waste decomposition, including CH_4 volumes and production rates.

A third effort[1] involves the effect of different waste processing techniques on gas and leachate production and duration during waste decomposition. Raw refuse, shredded refuse, and baled refuse are being investigated in a simulated landfill environment. Interim results of this effort have been previously discussed at the

Engineering Foundation and ASCE Conference: Land Application of Residual Materials, held in Easton, Maryland, September 26 to October 1, 1976, as "A Study of Gas and Leachate Production from Baled and Shredded Municipal Solid Wastes."

POLLUTANT TRANSPORT

Pollutant transport studies involve the release of pollutants in liquid and gaseous forms from various municipal and hazardous wastes and the subsequent movement and fate of these pollutants in soils adjacent to disposal sites. Although the potential for damage in general can be demonstrated, migration patterns of contaminants and consequent damages that would result from unrestricted landfilling at specific sites cannot be accurately predicted. The ability to predict must be developed in order to justify the requirement for changes in the design and operation of disposal sites, particularly for any restriction of co-disposal of municipal and industrial waste. Both laboratory and field verification studies at selected sites are being performed to assess the potential for groundwater contamination. The studies will provide the information required to (1) select land disposal sites that will naturally limit release of pollutants to the air and water and (2) make rational assessments of the need for and cost-benefit aspects of leachate and gas control technology.

The overall objective of this research activity is to develop procedures for using soil as a predictable attenuation medium for pollutants. Not all pollutants are attenuated by soil, and, in some cases, the process is one of delay so that the pollutant is diluted in other parts of the environment. Consequently, a significant number of the research projects funded by SHWRD are focused on understanding the process and predicting the extent of migration of contaminants (chiefly heavy metals) from land disposal sites.

These pollutant migration studies are being performed simultaneously in the areas of municipal refuse and specialized wastes. Several previous reviews of these efforts have been presented. See Roulier, M. H., Research on Minimizing Environmental Impact from Landfilling, and Research on Contaminant Movement in Soils, both presented at a meeting of the NATO Committee for Challenges to Modern Society, Project Landfill, October 22-23, 1975, London, England.

Bibliography and State of the Art

A state-of-the-art document on migration through soil of potentially hazardous pollutants contained in leachates[5] from waste materials has been published, Movement of Selected Metals, Asbestos, and Cyanide in Soils: Applications to Waste Disposal Problems, EPA-600/2-77-022, April 1977. The document presents a critical review of the literature pertinent to biological, chemical, and physical reactions, and mechanisms of attenuation (decrease in the maximum concentration for some fixed time as distance traveled) of the selected elements arsenic, beryllium, cadmium, chromium, copper, iron, mercury, lead, selenium, and zinc, together with asbestos and cyanide, in soil systems.

Controlled Lab Studies

The initial effort[5] is examining the factors that attenuate contaminants (limit contaminant transport) in leachate from municipal solid waste landfills. Although the work is strongly oriented towards problems of disposal of strictly municipal wastes, the impact of co-disposal of municipal and hazardous wastes is also considered. The project is concerned with contaminants normally present in leachates from municipal solid waste landfills and with contaminants that are introduced or increased in concentration by co-disposal of hazardous wastes. These contaminants are: arsenic, beryllium, cadmium, chromium, copper cyanide, iron, mercury, lead, nickel, selenium, vanadium, and zinc. The general approach was to pass municipal leachate through columns of well characterized whole soils maintained in a saturated anaerobic state. The typical municipal refuse leachate was spiked with high concentrations of metal salts to achieve a nominal concentration of 100 mg/l. The most significant factors in contaminant removal were then inferred from correlation of observed migration rates and known soil and contaminant characteristics. This effort will contribute to the development of a computer simulation model for predicting trace element attenuation in soils. Modeling efforts to date have been hindered by the complexity of soil-leachate chemistry. Interim results of this effort have been reported by Fuller, W. H., et al., 1976, "Contribution of Soil to Migration of Certain Common and Trace Elements," Soil Sci. 122: 223-235 and by Korte, N. E., et al., 1976, "Trace Element Movement in Soils: Influence of Soil Chemical and Physical Properties", Soil Sci. 122: 350-359.

The second effort[5] in this area is studying the removal of contaminants from landfill leachates by soil clay minerals. Columns were packed with mixtures of quartz sand and nearly pure clay minerals. The leaching fluid consisted of typical municipal refuse leachate without metal salt additives. The general approach to this effort was similar to that described in the preceding effort except that (1) both sterilized and unsterilized leachates were utilized to examine the effect of microbial activity on hydraulic conductivity and (2) extensive batch studies of the sorption of metals from leachate by clay minerals were conducted. Interim results of this effort have been reported in two articles by Griffin, R. A., et al., "Attenuation of Pollutants in Municipal Landfill Leachate by Clay Minerals: Part 1 - Column Leaching and Field Verification," Environmental Geology Notes, No. 78, November 1976, Illinois State Geological Survey, Urbana, Illinois, and "Attenuation of Pollutants in Municipal Landfill Leachate by Clay Minerals: Part 2 - Heavy Metal Adsorption Studies," Environmental Geology Notes, No. 79, March 1977, Illinois State Geological Survey, Urbana, Illinois.

The third effort[4] relates to modeling movement in soil of the landfill gases, carbon dioxide and methane. The modeling movement has been verified under laboratory conditions. This effort has not focused on the impact of gases on groundwater, but considers groundwater as a sink for carbon dioxide. Results to date have involved design curves and tables which have been used to successfully evaluate a gas problem in Minnesota.

A fourth effort[5] relates to the use of large-scale, hydrologic simulation modeling as one method of predicting contaminant movement at disposal sites. The two-dimensional model that was used successfully to study a chromium contamination problem is being developed into a three-dimensional model and will be tested on a well-monitored landfill where contaminant movement has already taken place. Although this type of model presently needs a substantial amount of input data, it appears promising for determining contaminant transport properties of field soils and, eventually, predicting contaminant movement using a limited amount of data.

Field Verification

Limited field verification is being conducted. The initial effort to date has consisted of installing monitoring wells and coring soil samples adjacent to three municipal landfill sites to identify contaminants and determine their distribution in the soil and groundwater beneath the landfill site. The sites represent varying geologic conditions, recharge rates, and age, ranging from a site closed for 15 years to a site currently operating. Individual site characteristics were identified, and sample analyses necessary to determine the primary pollutant levels in the waste soils and groundwater were determined. Validation of waste leachability and pollutant migration potential are to be determined.

Organic Contaminants

The initial effort[1] relates to organic contaminant attenuation by soil. Much more is known about inorganic contaminant movement in soil because the analytical techniques for inorganic materials are well developed and relatively cheap compared to the time-consuming analytial techniques for organic materials. The problem is compounded by the fact that organic contaminants are more numerous and more are being synthesized all the time. PCB is the organic contaminant currently being investigated. In initial results, PCB's were found to be immobile in earth materials when measured by the soil thin-layer chromatography technique.

POLLUTANT CONTROL

Pollutant control studies are needed because experience and case studies have shown that some soils will not protect groundwater from contaminants. Even sites with "good soils" may have to be improved to protect against subsurface pollution. The overall objective of the pollutant control studies is to lessen the impact of pollution from waste disposal sites by technology that minimizes, contains, or eliminates pollutant release and leaching from waste residuals disposed of to the land.

The pollutant control studies are determining the ability of in-situ soils and natural soil processes to attenuate leachate contaminants as the leachate migrates through the soil from landfill sites. The studies are also determining how various synthetic and admixed materials may be utilized as liners to contain and prevent leachates from migrating from landfill sites.

Natural Soil Processes

The treatment by natural soil processes of pollutants from hazardous waste and municipal refuse disposal sites is being performed in Pollutant Transport studies wherein various soils are being evaluated in column studies for their pollutant attenuation capabilities. The initial effort[5] is current investigation of soils ranging in texture from sands to clays. In a second effort[5] various percentages of the clay minerals kaolinite, montmorillonite, and illite are mixed with pure sand to form various mixtures that are packed into columns for study. A state-of-knowledge report describing attenuation mechanisms has been published. See Movement of Selected Metals, Asbestos, and Cyanide in Soils: Applications to Waste Disposal Problems, EPA-600/2-77-022, April 1977.

Liners/Membranes/Admixtures

The liner/membrane/admixture technology[6] is being studied to evaluate suitability for eliminating or reducing leachate from landfill sites of municipal or industrial hazardous wastes. The test program will evaluate, in a landfill environment, the chemical resistance and durability of the liner materials over 12-, 24-, and 36-month exposure periods to leachates derived from industrial wastes, SO_x wastes, and municipal solid wastes. Acidic, basic, and neutral solutions will be utilized to generate industrial waste leachates.

The liner materials being investigated under the municipal solid wastes program include six admixed materials and six flexible membranes. The admixed materials are:

- 2 asphalt concretes, varying in permeability
- 1 soil asphalt
- 2 asphalt membranes, one based on an emulsified asphalt and the other on catalytically-blown asphalt
- 1 soil cement

The six flexible membranes are:

- Butyl rubber
- Ethylene propylene rubber (EPDM)
- Chlorinated polyethylene (CPE)
- Chlorosulfonated polyethylene (HYPALON)
- Polyethylene (PE)
- Polyvinyl chloride (PVC)

Specimens of these 12 liner materials have been exposed for more than 2 years to landfill leachates generated in individual simulated landfills. After 1 year of exposure, a set of 12 of the simulated landfills was dismantled and the liners retrieved and tested. The results of the 1-year exposure have been discussed in a report entitled Evaluation of Liner Materials Exposed to Leachate - Second Interim Report, EPA-600/2-76-255, September 1976.

POLLUTANT TREATMENT

The pollutant treatment studies relate to the collected leachate that is physically, chemically, or biologically treated prior too discharge from the landfill site. Also, recirculation and spray irrigation concepts are considered to be potential treatment schemes. The overall objective of the pollutant treatment studies is to develop technology that treats the landfill leachate once it has been collected and contained at the landfill site.

Physical-Chemical Treatment

Various physical-chemical treatment schemes[4] were investigated in the laboratory. Physical-chemical treatments consisted of chemical precipitation, activated carbon adsorption, and reverse osmosis. The activated carbon was quite effective in removing refractory organics in the effluent of biological units. The most promising treatment scheme, an anaerobic lagoon followed by aerobic polishing, was selected for pilot plant evaluation.[1] The results of this initial effort have been reported by Ho, S., Boyle, W. C., and Ham, R. K., "Chemical Treatment of Leachates from Sanitary Landfills," JWPCF, Vol. 46, No. 7, July 1, 1974, pp. 1776-1791.

A second effort[4] on the physical-chemical treatment schemes was an expansion of the initial effort not only in the areas of chemical precipitation, activated carbon, and reverse osmosis, but also in ion exchange,adsorption and chemical oxidation. The results of this effort have been reported by Chian, S. K. and DeWalle, F. B., "Sanitary Landfill Leachates and Their Treatment," JEED, ASCE, Vol. 102, No. EE2, Proc. Paper 12033, April 1976, pp. 411-431.

A third effort[5] involves a laboratory evaluation of various materials that could be utilized as retardant materials to minimize migration of pollutants from disposal sites. This investigation involves study of the following materials on a pilot plant basis: agricultural limestone, hydrous oxides of Fe (ferrous sulfate mine waste), lime-sulfur oxide (stack-gas waste), certain organic wastes, and soil sealants. Preliminary research on limestone and Fe hydrous oxide liners indicates these materials have a marked retarding influence on many of the trace elements. However, the increased water contamination from solubilization of iron seems to rule out use of iron oxides in this treatment scheme.

Biological Treatment

Various unit processes for biological treatment of leachate[4] have been investigated in the laboratory. The results of this initial effort have been reported by Boyle, W. C. and Ham, R. K, "Treatability of Leachate from Sanitary Landfills," JWPCF, Vol.46, No. 6, June 1974, pp. 860-872. A second effort[4] has investigated the process kinetics, the nature of the organic fraction of leachate, and the degree of treatment that may be obtainable using conventional wastewater treatment methods. The biological methods evaluated were the anaerobic filter, the aerated lagoon, and combined treatment of activated sludge and municipal sewage. Biological units were operated successfully without prior removal of the metals that were present in high concentrations. The results of this effort have been reported by Chian, S. K. and DeWalle, F. B., "Sanitary Landfill Leachates and Their Treatment," JEED, ASCE, Vol. 102, No. EE2, Proc. Paper 12033, April 1976k pp. 411-431.

Recirculation

Recycling of leachate[4] is being investigated to determine the beneficial aspects of recirculation as a means of leachate control and accelerated landfill stabilization. Recommended design, operation, and control methods applicable to conventional sanitary landfill practice will be developed. This effort was discussed earlier under the "Waste Decomposition" section, and the published report is mentioned in that section.

Spray Irrigation

Spray irrigation[4] of leachate is being investigated as a low-cost, onsite treatment scheme. Optimum leachate loading rates and removal efficiencies for organic and inorganic constituents are being determined for two soil types. The technique appears to be sensitive to moisture stresses (drought).

CO-DISPOSAL

In an effort to assess the impact of co-disposal, the disosal of industrial waste materials with municipal solid waste, a project utilizing large scale experimental landfill test cells was undertaken. Concern has been voiced that the addition of industrial waste may result in the occurrence of various toxic elements in leachates and thereby pose a threat to potable groundwater supplies. Because the environmental effects from landfilling result from not only the soluble and slowly soluble materials placed in the landfill but also the products of chemical and microbiological transformations, these transformations should be a consideration in management of a landfill to the extent that they can be predicted or influenced by disposal operations.

Presently, little is known on what effect adding industrial waste has on the decomposition process and the quantity and quality of gases and leachate produced during decomposition. There is a strong concern that addition of industrial wastes, particularly those high in heavy metals, will result in elevated metal concentrations in the leachates and potentially, in potable groundwater supplies. Advocates of co-disposal of sludges and municipal waste believe the presence of organics in the landfill will immobilize heavy metals. They also believe the presence of such sludges may accelerate the decomposition process and shorten the time required for biological stabilization of the refuse. Because of the high moisture content and, commonly, the high pH and alkalinity of these sludges, periodic analyses of the leachates in this study for trace and heavy metals is expected to provide data to allow rational evaluation of the practice of co-disposal. The overactivity is to evaluate and develop a predictable formulation for the transformation process.

The initial effort[4] involves a study of the factors influencing (1) the rate of decomposition of solid waste in a sanitary landfill, (2) the quantity and quality of gas and leachate produced during decomposition, and (3) the effect of admixing industrial sludges and sewage sludge with municipal refuse. A combination of municipal solid waste and various solid and semi-solid industrial wastes was added to several field lysimeters. All material flows were measured and characterized for the continuing study and related to leachate quality and quantity, gas production, and microbial activity.

The industrial wastes investigated were: petroleum sludge, battery production waste, electroplating waste, inorganic pigment sludge, chlorine production brine sludge, and a solvent-based paint sludge. Also, municipal digested primary sewage sludge dewatered to approximately 20 percent solids was utilized at three different ratios. The results of this initial effort have been reported by Streng, D. R., "The Effects of Industrial Sludges on Landfill Leachate and Gas," Proceedings - National Conference on Disposal of Residues on Land, September 1976, pp. 69-76.

A second effort[5] to assess the potential effects of co-disposal involves the leaching of industrial wastes with municipal landfill leachate as well as water. Results to date indicate that, when compared with water, municipal landfill leachate solubilizes greater amounts of metals from the wastes and promotes more rapid migration of metals through soil. The municipal landfill leachate is a highly odorous material containing many organic acids and is strongly buffered at a pH of about 5. Consequently, it has proved to be a very effective solvent. It is anticipated that during the life of this effort, studies will be conducted on 43 industrial wastes, 3 types of coal flyash, and 6 sludges generated by the removal of sulfur oxides from the flue gases of coal-burning power plants. Because of the difficulty in handling and analyzing municipal landfill leachate, it will be used with only some of these wastes.

A third effort[6] involves a study of the effects of co-disposing of chemically stabilized sludges in a municipal refuse landfill. This effort has just been initiated and the simulated landfill lysimeters have recently been constructed. It is anticipated that the loading of these test lysimeters will be completed by August 1977.

ENVIRONMENTAL ASSESSMENT

The environmental effects of waste disposal to the land need to be determined in relation to the management and disposal practices for municipal solid wastes. In an effort to assess the impact of these practices, several studies have been initiated. The overall objective is to develop predictive procedures for forecasting adverse environmental effects from land disposal activities and to provide user documents for implementing field practices for those methods that eliminate or minimize adverse effects.

The initial effort[7] involves determining the effects of application of composted municipal wastes and sewage sludge on selected soils and plants of croplands. Multiple applications of composted municipal refuse totaling 900 metric tons per hectare have resulted in satisfactory crop growth with only a moderate increase of some heavy metals in plant tissues. Very little downward movement of heavy metals was observed under conditions of heavy leaching in greenhouse or natural outdoor conditions. The effort has been published in a USEPA report entitled Effect of Land Disposal Applications of Municipal Wastes on Crop Fields and Heavy Metal Uptake, EPA-600/2-77-014, April 1977.

A second effort[6] involves a determination or evaluation of vegetation kills and growth problems associated with landfill gas migration as evaluated by mail survey and on-site investigations. Additional investigations are being performed to determine control measures for reducing vegetation losses, and experimental plot observations should determine those vegetation species most conducive to landfill environs. The results of this initial effort have been reported by Flower, F. B., Leone, I. A., and Gilman, E. F., "An Investigation of the Problems Associated with Growing Vegetation on or Adjacent to Refuse Landfills," Proceedings Physical Environment Conference, August 1975.

A third effort[1] has recently been initiated to study the operational and aesthetic effects of milled refuse particle size in a landfill operated without daily cover. The overall objective of this research is to establish acceptable parameters for the operation and maintenance of milled refuse landfills in order to minimize detrimental environmental effects. Specific variables to be evaluated are: the effect of wind velocities and direction on the movement of landfilling material; the amount of differential settlement associated with particle size variations; the initial density in each test cell and subsequent density with relation to time and consolidation within the cell and the presence or absence of surface crusting; qualitative evaluation of nuisance organisms, wildlife, and the type and amount of plant growth; and some evaluation of odors and background conditions potentially responsible for noticeable odors.

REMEDIAL ACTION

OSW has concluded the investigation of 391 damage cases. Fifteen percent of these cases involved groundwater pollution from hazardous waste landfills, 25 percent involved groundwater pollution from indiscriminate dumping practices, and 40 percent involved leachate problems. Nine percent or 35 of the 391 damage cases involved well pollution. An ongoing study by OSW has identified 50 incidents of well contamination due to municipal landfill disposal sites. Seventy-five to 85 percent of all MSW sites investigated are contaminating ground or surface waters. In order to determine the best practical technology and economical corrective measures to remedy these landfill leachate and gas pollution problems, a research effort has recently been initiated[2] to provide local municipalities and users with the data necessary to make sound judgments on the selection of viable, in-situ, remedial procedures and to give them an indication of the cost that would be associated with such a project. This research effort consists of three phases. Phase I will be an engineering feasibility study that will determine on a site specific basis the best practicable technology to be applied from existing neutralization or confinement techniques. Phase II will determine the effectiveness, by actual field verification, of the recommendations/first phase study. Phase III will provide a site remedial guide to local municipalities and users.

LANDFILL ALTERNATIVES/LAND CULTIVATION

Municipal solid wastes are primarily deposited in standard sanitary landfills or incinerated. Because of concern for environmental impact and economics, other landfill alternatives have been proposed.

For SHWRD purposes, the alternatives currently being considered are: (1) deep well injection, (2) underground mines, (3) land cultivation, and (4) saline environments. The deep well injection and underground mine alternatives are strictly orientated towards hazardous wastes and as such will not be discussed in this symposium or its proceedings. The remaining two alternatives are, however, orientated towards municipal waste. In order to assess the feasibility and beneficial aspects of spreading and admixing municipal refuse into the soil, an initial effort[6] involves land cultivation and the preparation of a state-of-the-art document. Available data indicate that application of shredded municipal refuse or compost to marginal or drastically disturbed land improves soil structure and fertility, thus making revegetation possible. It appears that the environmental pollution caused by land cultivation is minimal as compared to that for landfills, primarily due to maintenance of aerobic conditions and the lower concentration of waste per unit area of land. Technical and economic assessment efforts will follow.

A second effort[1] recently begun involves documenting the disposal of municipal solid wastes in saline environments, i.e., estuaries and coastal marshlands. The purpose of this study is to obtain a document detailing the present environmental and economic status of municipal solid waste disposal into specific saline environments and compiling state regulations and policies in effect for those states bordering saline waters.

ECONOMIC INCENTIVES

The use of market-related incentive (disincentive) mechanisms has received only scant consideration for pollution control policy in the United States, particularly in the area of solid waste management. It has been hypothesized that incrementally pricing solid waste collections and disposal would reduce the waste generation rate, enhance source separation of recyclable materials, accelerate technological innovation, and minimize total system cost. It has also been asserted that properly structured prices or user charges would be an equitable means of allocating public resources and an efficient system for maximizing net social benefits of a municipally provided service. In order to evaluate these statements, award of the initial effort[8] to delineate the conditions under which user charges would be economically feasible for solid waste management is anticipated in FY 1977.

CONCLUSION

The laboratory and field research project efforts discussed here reflect the overall SHWRD effort in municipal solid waste disposal research. The projects will be discussed in detail in the following papers. More information about a specific project or study can be obtained by contacting the project officer referenced in the text. Inquiries can also be directed to the Director, Solid and Hazardous Waste Research Division, Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, 26 West St. Clair Street, Cincinnati, Ohio 45268. Information will be provided with the understanding that it is from research in progress and that conclusions may change as techniques are improved and more complete data become available.

PROJECT OFFICERS

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SUMMARY OF OFFICE OF SOLID WASTE GAS AND LEACHATE ACTIVITIES

Truett V. DeGeare, Jr.
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Two years ago this month I had the pleasure of participating in the first of this series of symposia being sponsored by our sister office, the Solid and Hazardous Waste Research Division. At that time I discussed several field projects, in the areas of landfill gas and leachate, in which we were involved or hoped to become involved. I also indicated the likelihood that in the future we would see a need for expansion of land disposal criteria or guidelines at the Federal and/or State levels. Today, I intend to again discuss some of our field projects, a few of which you may recognize as being only desires two years ago. I also, again, intend to briefly discuss the need for expansion of disposal criteria or guidelines, an area of anticipation only two years ago which today is reality due to enactment of the Resource Conservation and Recovery Act of 1976.

Field Projects

Leachate Control

In the area of leachate control we are currently involved in five significant efforts; four of which relate to leachate treatment, while the fifth is concerned with containment.

At Enfield, Connecticut, an anaerobic filter system has been constructed for the treatment of leachate. The design was based on bench-scale research conducted by the University of Illinois. The project was instituted in June 1975 with bench-scale studies and has proceeded through design and construction of the facility. Construction is complete; however, system start-up has been delayed pending mechanical repair found necessary during testing. The bench-scale testing has indicated that we may expect COD removals of up to 90%. Evaluation of the field facility will be conducted through April 1978.

In Falls Township, Pennsylvania, Waste Resources Corporation has constructed a physical/chemical/ biological treatment facility to treat leachate generated by its lined landfill. Although we were not involved in the construction of this facility, we have been funding an evaluation of the plant's operation with regard to cost and performance. The evaluation is being conducted by Applied Technology Associates. This plant was of special interest to us for evaluation in that its design provided for variation in mode of operation. Thus, at one facility and with a single source of leachate, we were able to evaluate four systems.

1. chemical/physical treatment followed by biological (activated sludge),
2. chemical/physical treatment only,

3. biological treatment followed by chemical/physical,

and

4. biological treatment only.

In addition, carbon absorption polishing of each of the above system effluents could be examined on a bench-scale basis. To date, systems 1 and 2 have been evaluated with encouraging results. Systems 3 and 4 have received only preliminary attention; however, it appears that the raw leachate inhibits the activated sludge process to the extent that pretreatment is required. These latter two systems will be evaluated further this spring. Thus far, the plant has performed most satisfactorily in the mode of system 1 which, for example, reduced BOD by 99%, COD by 94%, cadmium by 71%, iron by 99%, and lead by 78%. Costs for this mode are in the range of $4 to $5 per thousand gallons treated. Further details on this project should be available this summer in an interim report which is currently being prepared.

Through an interagency agreement we are contributing to the USDA Agricultural Research Service's evaluation of the performance of a leachate treatment system. This system, located near Bluefield, West Virginia, employs an aerated lagoon and spray irrigation. Leachate from the lagoon is being sprayed on plots of vegetated land from which soil, soil moisture, and vegetation samples are obtained and analyzed. The study is intended to determine:

1. The efficiency of spray irrigation as a decontamination method,

2. The effects of lime treatment of soil on the soil's ability to complex and retain leachate contaminants,

3. The effectiveness of the aerated lagoon, and

4. The effects of spray irrigation on the vegetation.

Results of this study are not yet available; however, an interim report is under preparation.

We have just recently contracted for the field evaluation of up to six existing leachate treatment systems. The recent Waste Age Survey of U.S. Disposal Practices (as reported in Waste Age, January 1977) indicated the existence of some 200 such facilities. Our intent here is to provide case study information on the design, cost, and performance of specific systems presently in use. Hopefully, this information will be of value to engineers and disposal facility operators who are considering the design and operation of leachate treatment facilities. The facilities to be evaluated have not yet been identified. Should you wish to suggest particular facilities for evaluation, I would like to hear from you by the end of the month.

As I mentioned in 1975, we are using funds provided by the Appalachian Regional Commission to support the construction of a lined sanitary landfill in Lycoming County, Pennsylvania. The project has moved through design to solicitation of construction bids. We are hoping that the facility will begin operation this fall. This project will provide full documentation of the design, construction techniques, and costs of a lined sanitary landfill. For additional information on the design of this facility, I suggest you review a recent paper on the Lycoming County project which was prepared by Todd Giddings and published in the January-February 1977 edition of Ground Water.

Leachate Monitoring

We are continuing several of the leachate monitoring efforts which were funded in earlier years. These continuations are significant in that, although leachate generation and movement are recognized to be time-dependent, long-term data are rare.

In 1970 we initiated a project to demonstrate a unique approach to sanitary landfill operation in a high water table area, namely, Orange County, Florida. Part of the demonstration involved the installation and monitoring of a series of ground water wells. The Orange County Pollution Control Board has continued to monitor this facility for seven years, an effort scheduled to terminate this fall. The data has indicated that subsurface leachate movement has been confined to the area directly beneath the completed cells; vertical movement has been detected to a depth of 27 feet. The surface drainage system, designed to prevent flooding and to lower the water table, has exhibited increased oxygen demand only in the detention pond and part of the outfall canal.

This summer will bring to an end Professor Robert Ham's seven-year monitoring effort at the University of Wisconsin. Dr. Ham will be producing a final report on a series of 400-cu-yd lysimeters constructed to simulate the land disposal of milled and unprocessed solid waste under various conditions.

During 1973 and 1974 we participated in the evaluation of a baler operation in St. Paul, Minnesota. The evaluation included the construction and monitoring of a test cell of baled solid waste; however, only short-term monitoring was provided through that project. A report on that project, conducted by Ralph Stone and Company, is available from our office. Last summer we were able to fund an effort by the Univerisity of Minnesota to repair the monitoring system, which had been unattended for two years, and reinstitute a monitoring program. Due to physical problems at the site, including a severe winter, little data has been obtained thus far. The project is scheduled to terminate in 1978.

In speaking of monitoring, I wish to mention a recent contract effort which produced a document which I think would be of value to many of you. In our discussion with researchers, consultants, site operators, and representatives of State and local government it had become obvious that guidance was needed and desired in the area of leachate monitoring, especially with regard to subsurface movement. In response to this need we contracted with Wehran Engineering Corporation and Geraghty and Miller, Inc. to produce a "Procedures Manual for Monitoring Solid Waste Disposal Sites." The manual was produced with valuable input from representatives of State agencies, the consulting engineering sector, U.S. Geological Survey, the disposal facility operators sector, and the water-well industry. The manual is very comprehensive, providing information on the objectives and costs of monitoring, specific monitoring techniques such as electrical earth resistivity and various types of well systems, sampling, and analytical techniques. The manual has been well received, and we have just recently asked for a second printing.

Gas

We are dealing with landfill gas in two areas: recovery and control.

Our gas recovery project is being conducted by the City of Mountain View, California, and Pacific Gas and Electric Company. The project is progressing, although slowly, into the production stage. The gas treatment unit has been constructed and the compressor has been ordered for June delivery. This spring the 18 production wells will be constructed. The compressor and molecular sieve treatment unit will then be installed and gas will be extracted, upgraded to 750 BTU/scf, and injected into an adjacent commercial gas pipeline. The system is to extract 1 million scf per day and produce about 500,000 scf per day of high quality gas. Reports by the City on gas extraction and by the utility on gas utilization should be available from us this summer.

Severe gas migration problems continue to come to our attention. However, there are few sources of information on control techniques. In order to provide guidance on control systems which have been used in the field either successfully or unsuccessfully, we have recently obtained a report from Engineering Science, Inc. on the design, construction, operation, and costs of five such control systems. In that we received the report only last week and have not had an opportunity to review it, I can't reflect further on its contents. We hope to have the report available for distribution this summer.

Criteria and Guidelines

As you are probably now all aware, last October the Resource Conservation and Recovery Act of 1976 (RCRA) became law. With regard to solid waste disposal, RCRA includes some significant definitions and requirements which I would like to discuss.

Three of the requirements placed on EPA are that we:

1. promulgate regulations containing criteria for determining which disposal facilities shall be classified as sanitary landfills or as open dumps;
2. publish an inventory of all disposal facilities in the country which are open dumps; and
3. promulgate suggested guidelines, including a description of levels of performance to protect ground waters from leachate.

RCRA recognizes "open dumps" and "sanitary landfills" as the only types of solid waste disposal facilities and defines them by reference to classification criteria which are to be promulgated under §4004. RCRA also specifically defines the terms "disposal" and "solid waste" with considerable breadth:

> The term "disposal" means the discharge, deposit, injection, dumping, spilling, leaking, or placing of any solid waste or hazardous waste into or on any land or water so that such solid waste or hazardous waste or any constituent thereof may enter the environment or be emitted into the air or discharged into any waters, including ground waters.

> The term "solid waste" means any garbage, refuse, sludge from a waste treatment plant, water supply treatment plant, or air pollution control facility and other discarded material, including solid, liquid, semisolid, or contained gaseous material resulting from industrial, commercial, mining, and agricultural operations, and from community activities, but does not include solid or dissolved material in domestic sewage, or solid or dissolved materials in irrigation return flows or industrial discharges which are point sources subject to permits under Section 402 of the Federal Water Pollution Control Act, as amended or source, special nuclear, or by-product material as defined by the Atomic Energy Act of 1954, as amended.

As I said earlier, RCRA defines the terms "open dump" and "sanitary landfill" by reference to specific criteria to be developed pursuant to §4004. This section requires that we promulgate regulations containing criteria for determining which facilities shall be classified as sanitary landfills and which shall be classified as open dumps. At a minimum, the criteria are to provide that a facility may be classified as a sanitary landfill and not an open dump only if there is no reasonable probability of adverse effects on health or the environment from disposal of solid waste at the facility. An important aspect of the implementation of RCRA, then, is delineation of what constitutes "no reasonable probability" and what constitutes "adverse effects on health or the environment." The law requires that these regulations be promulgated by October 21, 1977, after consultation with the States and after notice and public hearings.

Not later than one year after promulgation of the criteria for classification of disposal facilities, we must publish an inventory of all disposal facilities in the United States which are open dumps.

The legislative intent appears to be not for Federal regulation of disposal, but for State control. The law requires that each State plan prohibit the establishment of open dumps and contain a requirement that all solid waste within the State be disposed of in sanitary landfills unless it is utilized for resource recovery. The State prohibition is to be effective six months after promulgation of the criteria or on the date of approval of the State plan, whichever is later.

In addition to the State prohibition, the law contains a Federal prohibition of open dumping when usable alternatives are available. Where entities can demonstrate that they are unable to utilize such alternatives, the State plan must establish a timetable or schedule for compliance. The compliance schedule must specify remedial measures, including an enforceable sequence of actions, leading to compliance with the prohibition within a reasonable time not to exceed five years from the date of publication of the inventory. If a State plan is not being pursued, the citizen suit provision of the law provides recourse to aggrieved parties.

As did the 1965 Act, RCRA requires that we publish suggested solid waste management guidelines which are mandatory only for Federal agencies and certain recipients of Federal financial assistance. The guidelines are to provide technical and economic descriptions of levels of performance of solid waste management practices. Areas to be addressed by the guidelines include appropriate methods and degrees of control that provide for protection of public health and welfare; protection of the quality of ground waters and surface waters from leachates; protection of the quality of surface waters from runoff through compliance with effluent limitations under the Federal Water Pollution Control Act, as amended; protection of ambient air quality through compliance with new source performance standards or requirements of air quality implementation plans under the Clean Air Act, as amended; disease and vector control; safety; and esthetics. As in the 1965 Act and its amendments, RCRA does not specify which solid waste management practices are to be addressed by the guidelines. To us it seems most appropriate to first direct our guidelines efforts to the land disposal of solid wastes and the disposal and utilization of municipal waste water treatment sludges. In determining other practices to address, we solicit your viewpoints.

In the near future, probably next month, you can expect to see in the Federal Register Advance Notices of Proposed Rulemakings (ANPRs) addressing the criteria and guidelines. These notices will not be draft or proposed criteria and guidelines. They will be an explanation of the requirements of the law along with questions or issues of concern in developing the criteria and guidelines to meet the requirements. The notices are our way of formally notifying the public of our intentions and soliciting their response to the questions and issues. I hope that you will feel free to respond as described in the notices and provide any information or suggestions you may have to contribute.

At this time, so that you might be able to identify them later, I would like to introduce three other representatives of our Agency. Dr. John Skinner is directing the activities of our Systems Management Division which is responsible for the criteria and guidelines I have mentioned. Mr. Sheldon Meyers directs the activities of the Office of Solid Waste and will address you at the banquet tonight. Mr. Morris Tucker is responsible for our solid waste program in our Region VII with headquarters in Kansas City. I hope that during the course of this symposium you will not hesitate to contact any of us and express your views regarding our activities and the various provisions of this new law. Thank you for your attention.

STATE OF MISSOURI SOLID WASTE MANAGEMENT ACTIVITIES

Robert M. Robinson, P. E., Director
Solid Waste Management Program
Division of Environmental Quality
Missouri Department of Natural Resources

The Solid Waste Management Program activities in Missouri are directed primarily toward controlling the disposal of wastes on land. Wastes which we do not regulate include mining and farming residues and liquid wastes. I will discuss the program activities in three areas: land disposal, hazardous waste and planning. This breakdown of activities corresponds to the organization of our central office program.

The Solid Waste Management Program is one of eight environmental programs located in the Department of Natural Resources' Division of Environmental Quality. The Department of Natural Resources was established by reorganization of state government in 1974. This organization of environmental programs allows for close coordination of regulation and surveillance of environmental facilities within our state.

A total solid waste staff of ten is located in our Jefferson City office. Field investigations and technical assistance are provided by regional office program staff located in six regional offices throughout the state. Approximately four man-years of work are provided annually on solid waste activities by the regional office program staff. With a total of 14 man-years of effort it is readily apparent that the state has a modest commitment of staff to solid waste activities. However, I must add that the same relationship of commitment of staff and resources is also true for other environmental programs in Missouri.

Land Disposal

The program's activities to control disposal of solid waste is based on the legislative authority provided in 1972. The Missouri Legislature enacted a law (Sections 260.200 to 260.245, RSMo., 1975 Supplement) requiring a permit to operate solid waste disposal areas and processing facilities. A date of June 30, 1973, was set by law for obtaining a permit. It was, of course, not possible in one year to adopt regulations for disposal areas and processing facilities, close approximately 480 open dumps and permit an adequate number of disposal areas. The program established a policy of requiring the closure of open dumps as permitted sanitary landfills became available. Our efforts were directed toward the more highly populated areas first. At the present time less than 100 open dumps are known to exist in the state. We have written commitments for closure of many of these dumps as several permitted sanitary landfills begin operation this coming spring.

Since 1973 the state has issued one hundred and forty-seven (147) operating permits for sanitary landfills, demolition landfills, transfer stations, processing facilities and special disposal sites. Permits are issued for the life of the area or facility. Approximately 82% of the state's population is presently served by permitted sites.

The permitting process includes preliminary site evaluation by our department. This site investigation is conducted as a voluntary service to provide the applicant some assurance that the disposal area will be considered suitable before funds are invested to develop engineering plans. Site investigations are made by a geologist with the department's Division of Geology and Land Survey and an engineer from our Regional Office Program. From their reports we advise the applicant whether or not the geology and other physical characteristics of the landfill site look suitable. The applicant then makes the decision whether to employ a consulting engineer to submit detailed engineering design plans and the application. The Solid Waste Management Program reviews the application and plans within approximately 45 days of receipt. The permit is either issued, denied or revision of plans requested.

The solid waste enforcement activities have been rather limited to date for the following reasons: our policy of not closing dumps until permitted facilities are available, most disposal area operators have complied with the law and regulations by department request or orders, and the legal assistance available from the Attorney General's office is in short supply due to budget constraints. At our request five legal actions have been filed by the Attorney General to obtain closure of an open dump or compliance with operating regulations for sanitary landfills. None of the cases have resulted in a court trial because satisfactory agreements and compliance were reached.

Routine inspection of permitted disposal areas and processing facilities and agreements for voluntary closure of open dumps are accomplished by the staff in the six regional offices. Permitted disposal areas and processing facilities are inspected an average of twice per year. If the regional office is unable to obtain a reasonable commitment for closure of an open dump or the operator of a landfill fails to improve the operation after notification, the Solid Waste Management Program is requested to take legal action. We have several alternative actions to take which include: issuance of a department administrative order, holding of an informal show-cause hearing, issuance of a revocation of permit and request for legal action by the Attorney General's office. The situation determines which type of action is taken.

Several of the department's legal actions have involved the discharge of leachate from non-permitted landfills. However, we have experienced very few problems with leachate generation at permitted sanitary landfills. The problems that have occurred can be attributed to poor operation, such as failure to provide daily cover and proper control of surface drainage. Several factors may have helped to control leachate problems in recent years. The State of Missouri has experienced below normal rainfall the past three years, whereas normally the rainfall is approximately equal to evaporation. The sanitary landfill regulations prohibit the acceptance of waste containing free moisture. Leachate collection and treatment is required as part of the landfill design where site conditions indicate leachate discharges are likely to occur. Monitoring wells are being required where leachate could possibly discharge from the site without being detected on the ground surface.

Gas generation has not proven to be a problem with permitted sanitary landfills in Missouri. Gases are generally vented through the cover material without special design. We only require special provisions for gas venting when the landfill is located relatively close to buildings.

Hazardous Waste

A statewide hazardous waste survey has been a major activity of the Solid Waste Management Program during the past two years. We were assisted in conducting the survey by the Mid-America Regional Council in the Kansas City area. Four hundred eighty-one industrial plants were surveyed. Projection of the survey data indicates that industry in Missouri is generating potentially hazardous waste at the rate of about one million metric tons per year. The survey report and state plan for hazardous waste management will be completed and published this year.

Legislation to regulate hazardous

waste has been introduced in the Missouri Legislature. The legislation was developed with the assistance of a large volunteer committee representing industry, public interest groups, environmentalists, agriculture and state and local governments. With this broad support for the legislation we believe there is a good chance it will pass this legislative session.

Our program was instrumental in the development of the St. Louis Industrial Waste Exchange operated by the St. Louis Regional Commerce and Growth Association. This was the first waste exchange to be established in the U. S. We do not participate in the operation of the St. Louis exchange, but encourage industries to list their waste with it. Information is also provided industry on the availability of waste recyclers, processing facilities and hazardous waste disposal areas. Three special disposal sites have been permitted in the state to accept hazardous wastes. Wastes are accepted at these sites only after information on the properties of the hazardous waste and handling procedures are submitted to the Solid Waste Management Program and we authorize acceptance. Unfortunately, these three sites are all located on the western side of the state. We hope to have a special disposal site permitted [illegible] the St. Louis area within a few mon[illegible]

The waste market for these special disposal sites is created because the state solid waste regulations for sanitary landfills prohibit the acceptance of hazardous wastes, bulk liquids, semi-solids, sludges containing free moisture, highly flammable or volatile substances, unexpended pesticide containers, pesticides, raw animal manure, septic tank pumpings, raw sewage sludge and industrial process sludges unless appropriate design criteria and operating procedures are specifically delineated in the approved engineering design plans. Only a few sanitary landfills other than the special disposal sites have received approval to accept some of the above listed wastes. We will not approve the operating practice of mixing liquid wastes, pesticides or heavy metal sludges with municipal waste in the sanitary landfill because of the increased potential for generating leachate and increasing the difficulty of treatment.

Planning

The third major program activity is planning. The solid waste law passed in 1972 and revised in 1975 requires cities over 500 population and certain counties to submit a solid waste management plan to the department for approval. Our program provides technical assistance to the cities, counties and regional planning agencies in development, review and approval of their solid waste management plans. Most of the state's twenty regional planning commissions have participated in the planning process, resulting in area-wide disposal solutions in many cases. More than 50% of the state's 114 counties are served by one landfill or transfer station. Several land fills are serving two or more counties. More than 200 cities have implemented collection services that serve all households, creating efficient collection systems and controlling the problems of promiscuous dumping. Our resource recovery planning efforts have been limited to a study of the markets available for recovered material in the state.

Our planned activities for the future include: updating the 1972 State Solid Waste Management Plan; accepting responsibility for hazardous waste control under the Resource Conservation and Recovery Act, if we are successful in obtaining passage of the Missouri Hazardous Waste Management Act being considered by the legislature; establishment of a solid waste training program; closure of the remaining non-permitted disposal areas in the state this year; and encourage through the planning process area-wide resource recovery systems in the metropolitan areas of the state.

Summary

In summary, the State of Missouri has implemented a comprehensive program to control the disposal of solid waste on land. Although leachate and gas generation has not been a major problem at permitted landfills, the program gives special attention to these potential problems in the review of engineering design plans.

The management of hazardous waste has been addressed by conducting a statewide survey and introducing hazardous waste legislation. Hazardous waste disposal is presently controlled by prohibiting their disposal at sanitary landfills unless special design and handling procedures are approved.

The Solid Waste Management Program provides technical assistance to local governments in the development, review and approval of solid waste management plans.

In the future we plan to increase the role of the program from primarily a land-disposal control agency to include resource recovery planning, solid waste training and comprehensive regulation of hazardous wastes.

REGION VII SOLID WASTE ACTIVITIES

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ABSTRACT

The states included within E.P.A., Region VII, are Iowa, Kansas, Missouri, and Nebraska. These four states have all expended a great amount of effort in developing and implementing solid waste management programs. From a minuscule total staffing of two man-years of effort available for this activity in 1968, the four states utilized 43 man-years in 1976. Most of the early effort included survey of municipal solid waste systems and disposal practices, development of state plans, obtaining needed legislation, development of rules and regulations, and establishing monitoring and enforcement activities. Probably the most outstanding example of the success of these efforts is in improved land disposal of municipal solid waste. In 1970, the region's population of 11,231,000 was served by about 2,400 open dumps and 27 sanitary landfills. On January 1, 1977, this populace was served by about 600 open dumps and 484 sanitary landfills. The percentage of population served by acceptable land disposal facilities increased from 10 percent in 1970 to 86 percent at the start of 1977.

I would like to start off by expanding upon the welcome to EPA, Region VII. It has already been extended to some degree, even though those of you that came from the East only made it within the Region by a couple hundred yards. You probably know that the Mississippi River is our boundary line with our Region V.

You are going to be spending about two and a half days at this meeting on research and development activities within EPA and by their contractors. Within this subject area you will be concentrating on gas and leachate from landfills receiving municipal solid waste. One note I would like to make here is that I notice on the program there was no distinction made as to whether we were talking about dumps or whether we were talking about sanitary landfills. However, our new legislation, the Resource Conservation and Recovery Act, calls for there to be developed a very clear distinction between the two and that dumps, as so defined, be closed within a certain timeframe. It also calls for a redefinition of sanitary landfill before this can be accomplished. Others have touched on this during the program and I am sure that it will be discussed more later, so I will not elaborate on that in more depth at this time.

I would like to digress from research and development views for a few minutes and concentrate on what I generally call implementation. Implementation is really the primary activity that those of us at the EPA regional level and at the state and local levels are involved in solid waste management. Hopefully, we utilize information gained by means of the various research and development studies. Still, the fact remains, there are tremendous quantities of waste to be disposed of daily whether or not you have available to you totally environmentally acceptable facilities.

You are all familiar with reports that

have been published nationally on the amounts and types of waste generated. Some of these figures include approximately 135 million tons per year of municipal waste, an additional 260 million tons per year of non-hazardous industrial waste, excluding mining, and over 7 million tons per year of sewage sludges. Then we get into the big numbers - approximately 700 million tons per year of agricultural waste and approximately 1.8 billion tons per year of waste from the mining industry. Something that has not been reported on so much, but you will see a great deal of in the near future, is the breakdown of the amounts of hazardous waste or potentially hazardous waste. Figures that we have prepared to date, from eight industry groups only, show that this figure is in the range of about 23 million tons per year.

What do these figures mean to us at the regional level? It is only fair to assume that our states, the four states within Region VII, generate their fair share. These states which include Iowa, Kansas, Missouri and Nebraska have a population of about 12 million people and, therefore, generate about 8 million tons per year of municipal-type waste. This may not sound like a lot of waste to some of you from the larger cities back in the East, but there are also problems associated with sparse populations and the difficulties of combining and establishing regional systems. Regional figures for our portion of the other types of waste are not as well known. However, as Mr. Robinson just indicated, Missouri, as have the remainder of our states, has just completed or is nearing completion of supplementary studies on industrial wastes. Hopefully, reports from all four of our states will be available this Spring.

Since one of the primary roles of EPA's regional offices is to translate the perceived national goals in approved waste management to the states, let's take a brief look at this activity. This essentially started with the passage in 1965 of the Solid Waste Disposal Act. This Act partially funded planning and training grants to assist our states and, in some cases, local and regional agencies to look at this somewhat new subject of improved solid waste management. At this time, and until about 1968, the four states within this region had a total of two man-years of effort devoted to solid waste management. In 1968, Missouri and Kansas accepted grants from EPA's predecessor agency to study the situation within their respective states and develop state plans. These states were followed in 1971 by Iowa and in 1972 by Nebraska. By this time, about 1972, the staffing for the four states had increased to 21 positions. Many of you are familiar with the activities conducted under these grant programs that involved essentially surveys of municipal systems and land disposal practices. Additionally, there was a great amount of training and technical assistance offered both by those of us in the regional office and by state personnel. State plans have been completed for all four of our states and generally effective legislation has been obtained to deal with municipal solid waste. All four states have been spending the bulk of their time since plan completion on implementation activities. This includes continued training together with increased efforts in monitoring and enforcement activities. Currently, the staffing for the four states in this region totals 43 positions for these programs. Additionally, over the past few years we have assisted in the development of about 15 regional plans through grants to local and regional agencies.

Probably one of the best measures of the effectiveness of this overall activity is in improved land disposal of municipal waste. Some summary figures on this shows that for 1970 the four states in the region, with a population base of about 11.2 million, had 2,400 open dumps, 27 approved sanitary landfills, and that 10 percent of the population were served by approved facilities. However, we think we have done some good since as of the first of this year, comparible figures show that we still have about 600 open dumps, most of them small, but we have increased the state-approved landfills to 484 and currently 86 percent of our populace is served by these approved facilities.

Robbie touched on the hazardous waste survey that has recently been done in the State of Missouri. Additionally, this has been done in the State of Kansas, which included about 450 firms, the State of Nebraska surveyed about 90 firms. I do not have any firm numbers on the State of Iowa but they have a comparable activity underway. As follow-up to these studies, all four of our states currently have legislative proposals before their legislators in an attempt to close the gaps in existing municipal solid waste legislation and to obtain needed additional legislation specifically for hazardous waste management.

I would like to touch just briefly on some of the research and development activities that have taken place within this region. Back in the early days of our program we had a demonstration project in Sarpy County, Nebraska, for sanitary landfill to reclaim a ravine. This project, although it doesn't sound so glamorous today, was highly successful and the reclaimed land now supports a bumper crop of corn each year. We subsequently had a large scale demonstration project with the Mid-Americal Regional Council (MARC) to establish a demonstration sanitary landfill project in Kansas City, Kansas. This project was unique in many aspects, possibly one of the greatest factors being that it was located immediately in an urban area. The landfill was bordered on two sides by residential housing within about 50 yards of the operation. Needless to say, there was substantial opposition when this was first started. However, appropriate orientation of local citizen leaders and neighbors convinced them that it would be to their benefit to wipe out a very ugly-looking ravine that had been used to dump trash and, in just a short time, end up with a community park. This project was completed about two years ago and that park development is now taking place. MARC has maintained continual monitoring since initiation of the project on leachate. They have a fully designed leachate underdrain monitoring system and also a series of gas wells that are monitored weekly.

The new Resource Conservation and Recovery Act of 1976 reemphasizes the position that the state should be the primary force in the control of health and environmental problems resulting from improper solid waste management practices and as the institutional catalyst for increasing the conservation and recovery of resources. Toward this end, EPA will provide increased technical and financial support so as to maximize state assumption of responsibilities under the act. The primary state tasks described in the Act include development and implementation of upgraded solid waste management plans, the establishment of hazardous waste management regulatory programs and the longer term goal of the elimination of all open dumps. The EPA regional offices will manage the state grant programs and technical assistance delivery systems. Additionally, should any of the states be unable to upgrade its own dump inventory survey system, the regional offices will be called upon to handle this activity. This also applies to a very substantial part of the new act on hazardous waste management, in that if the states are unable to operate a comprehensive program for hazardous waste management, this activity is to be handled by the EPA regional offices, including the operation of a permit program.

LANDFILL RESEARCH ACTIVITIES IN CANADA

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ABSTRACT

In Canada, municipal and industrial solid waste management, including land disposal, is generally a private sector or local government responsibility carried out under enabling Provincial Government legislation and regulation. The Federal government's role is principally to develop solid waste management technology, to generate and evaluate technical solid waste management information, and to develop new or improved guidelines or codes of good practice relating to various waste management aspects. These activities are conducted on a national basis as a service and as a back-up to the public and to the regulatory agencies.

Current research activities relating to land disposal are underway in each of the following study areas:

a) Site evaluation and environmental impact assessment;
b) Landfill leachate migration and attenuation;
c) Landfill leachate control;
d) Standard procedures development;
e) Special wastes disposal on land; and,
f) Industrial waste management.

The objectives of this land disposal research program are to generate and evaluate technical information on each of the above study categories which can be either utilized in the preparation of specific technical guidelines or transferred to regulatory agencies for their direct consideration. The intent is to further develop techniques and procedures for improving waste management systems and enhancing environmental protection technology.

This paper describes Fisheries and Environment Canada's research projects which are currently in progress as part of the solid waste program on landfill, landfill leachate, and industrial waste disposal research.

INTRODUCTION

Research activities on solid waste are carried out in Canada by a number of Provincial Government agencies, a variety of research institutes and university groups, and a multitude of industries, in addition to the Federal Government.

This paper describes land disposal research activities carried out by the Federal Government, and in particular the Department of Fisheries and the Environment.

The Waste Management Branch is currently undertaking a number of research activities related to the discharge of both municipal and industrial or hazardous wastes onto land, in recognition of the need to improve existing control or regulatory measures in order to ensure proper and safe disposal of wastes onto land.

Some of our responsibilities in this regard are to develop technology, to provide regulatory agencies in Canada with technical information as a back-up service to them, to develop new or improved disposal guidelines, and to develop methods for predicting and monitoring environmental impacts of proposed waste disposal schemes. The results of our own land disposal research program along with research findings of others, will be used accordingly. At the same time, our research findings are also being made available to a NATO/CCMS Pilot Study on Disposal of Hazardous Wastes (1), in which Canada is actively participating along with the U.S. and several European countries.

Our current research activities pertaining to land disposal are underway in each of the following study areas:

a) site evaluation and environmental impact assessment;
b) leachate migration and attenuation;
c) leachate control;
d) standard procedures development;
e) special waste disposal; and,
f) industrial waste management.

These activities are conducted on a national basis through research contracts administered by either this Branch or our Regional office representatives, often in co-operation with other federal or provincial government organizations.

In this paper, some of our landfill and landfill related research projects will be briefly discussed.

LANDFILL STUDIES

Monitoring and Modelling Studies

Five landfills in southern Ontario were the subject of a special in-depth investigation of both the magnitude of contaminant loadings to receiving streams and the contaminant attenuation afforded by the soils at the landfills.

The landfill sites vary in size from 6 acres to 78 acres (2.4 to 31.6 hectares) and contain from 6 million to 150 million cubic feet (0.17 to 4.3 million cubic meters) of waste. Several of the landfills have received a large variety of domestic, commercial and liquid industrial wastes.

For each of the five sites, the contractor prepared a complete description of the site; provided instrumentation and sampling to examine groundwater contamination; collected groundwater and soils data; and, prepared summary reports. Laboratory water quality analysis work was performed by Environment Canada.

The instrumentation of the study sites was designed to yield information for soil and groundwater characterization and for ground water flow analysis.

Wells were drilled at each of the five sites, and drill logs were prepared during the drilling operation so as to adequately identify the various soil strata encountered. All groundwater elevations were recorded.

Soil samples were taken from the major soil strata encountered. For each landfill site, the major soil types encountered were identified by sieve analysis. Sufficient amounts of each of the 5 soil types at the five sites were collected for subsequent laboratory soil analyses.

The wells were developed as monitoring points to allow extraction of groundwater samples for chemical analysis. To achieve this, they were fitted with well screens attached to plastic or steel pipe. In addition, some screens were sealed such that the wells could function as piezometers to yield data on groundwater movement.

The monitoring wells were strategically located at each site as nests of two wells apiece. The piezometers were placed within the saturated zone in coarse-grained soils, where possible.

All sites required a well to be placed into the buried refuse to the base of the fill to yield samples of raw leachate. All wells were adequately flushed after placement to remove contaminants introduced during drilling and installation. They were capped but perforated to prevent pressure build-up within the well.

The groundwater monitoring program required the measurement of water levels and conductivity, and the withdrawal of groundwater samples for chemical analysis. Water levels and conductivity readings were taken at all wells with the former expressed as elevations in feet above sea level. After these measurements were taken, the wells were flushed or bailed to waste a volume of water equal to at least twice that standing in the well.

Samples were then taken from all wells, and field parameters measured. The samples were further prepared, including filtration when necessary, and transported by the contractor to the laboratory for analysis. A total of 35 parameters including PCB's were chosen at the outset, however this list was reduced after a screening analysis of the first set of results. The data on water levels, conductivity and water quality were collected and reported by the contractor.

The hydrometeorological data required for the study period included precipitation, temperature, and evaporation. To monitor water infiltration and surface discharges of leachate, a number of lysimeters and weirs were also installed wherever necessary.

Data on soil characterization was required for identifying and classifying the various soils encountered. The specific tests which were conducted provided information about the chemical and physical properties of the soil necessary to make this classification.

The manner in which the soils and contaminants from the leachate interact specifically with respect to contaminant absorption (and/or desorption) was of primary interest in this study. While the study had as its major objective the identifying of the magnitude of contaminant flux, it was also considered important that the conditions under which the migration occurs be documented so as to enable the development of explanatory or predictive migration models. The data collected during soil characterizations were inputs for this purpose.

The specific laboratory soils analysis which were conducted included the following:

(a) grain size analysis to allow classification of soil samples and to enable comparison between samples using a standard system. Smaller sized particles tend to be more active in soil-leachate interactions and thus classification of soil into sizes was required to examine this condition;

(b) permeability measurement provided information necessary to construct models for groundwater movement. Whereas it was desirable that undisturbed samples be used for these analyses in the coarser soils, undisturbed samples were difficult to obtain. It was also thought to be the case that, because of anisotrophy, permeability measured in the direction of drilling may not be equivalent to the horizontal permeability, and therefore field permeabilities were measured through a falling head test;

(c) clay minerology, since it appears that clay size particles are the most active in soil-leachate interactions. The type of clay is also significant and thus an analysis of soils using X-ray defraction techniques was also conducted;

(d) soil organic matter, which may also be very active in contaminant attenuation;

(e) soil cation exchange capacities provided information specific to reactions between the soil and contaminants from migrating leachate. The information will be of use when explaining the removal or release of cation contaminants during the passage of leachate through soil;

(f) soil resident ions, and total soil chemistry analysis represents a means whereby the adsorption and fixation of contaminants may be determined. By determining resident ion content and total soil chemistry, both before and after exposure to leachate, differences may be attributed to adsorption and fixation (and/or desorption and dissolution) of leachate contaminants; and,

(g) organic carbon measurements which will be used in a manner similar to that described for the resident ion analysis.

The results showed that four of the landfills were found to be situated on non-productive lands (former pits and swamps) near major rivers and any deleterious quality effects are restricted to shallow, local groundwater flow systems and to local surface water courses. Soil attenuation is minimal at these sites and leachate discharges occur to nearby streams. The other larger landfill, selected especially for the collection of data for attenuation studies, is situated in an area where a large, shallow, groundwater flow system can be influenced. Groundwater moves about 3000 feet through surficial sand and gravel before discharging to streams.

All data collected, including groundwater stage and quality measurements, water infiltration and other climatic data, has been tabulated in a final report (2). This report also includes descriptions of the hydrogeology of each site.

The data is currently being evaluated in a follow-up study which will attempt to quantify contaminant loadings from the landfills and which will also attempt to numerically simulate contaminant migration behavior utilizing both a finite elements model and a modified mass balance model patterned after Elzy's work at the University of Oregon (3). The results of this work indicates that with the data on land, a predictive ground water contaminant migration model for use in extrapolating data to predict future contamination migration at these and other landfills will be successfully developed. This development will be documented in the final report.

As a part of this contract, a state-of-the-art document on ground water contaminant migration modelling has also been prepared (4).

Leachate Plume Migration and Attenuation

A much more in-depth study of leachate migration and attenuation was initiated at the C.F.B. Camp Borden landfill, which is situated within a simple hydrogeological setting. Such an almost idealized, simple setting was thought to be conducive to testing new approaches and techniques, and also thought to offer an opportunity to apply and evaluate numerical plume migration simulation methods.

The landfill is situated approximately 60 miles north of Toronto within a lowland physiographic unit. Most of the area falls within a flat to gently undulating "Camp Borden sand plain". For the most part, this basin was at one time part of a lake floor and its surface sediments are therefore of deltaic and lacustrine origin. These loose granular materials have been, in general, well drained by the entrenchment of the rivers of the area, all of which drain to the north, to Georgian Bay.

This 35-year-old, 14-acre landfill appears to have been constructed in a minor ravine that had been cut into the regional sand plain.

Boreholes had been previously drilled through the fill and around its boundaries. These early holes were investigated to maximum depths of 50 feet below surface and encountered extremely uniform subsoil conditions. Each of the borehole logs showed stratified, fine to medium textured, uniform sands. The size gradation curves visually demonstrated this uniformity. The coefficient of permeability for the sands was estimated to be in the order of 10^{-3} cm/sec.

A regional evaluation using water well records showed that the sands below the garbage probably extend to depths slightly more than 100 feet at the site. Below this, well records reveal a fairly thick sequence of stratified lacustrine silts and clays that in turn may be underlain by glacial tills of this same texture. These soils beneath the surface sands then probably act as an aquitard due to their low permeability. In effect then, the surface sands were thought to be sealed hydraulically at depth. Bedrock lies about 200 feet below the present land surface buried by the thick soil sequence.

Groundwater appears to flow in a northerly direction, and laterally within the surface sands. Thus precipitation infiltrates the landfill in nearly a vertical direction, and any contaminants that are leached from the solid waste migrate with this ground water flow. The site receives approximately 6.7 inches of average annual water surplus, based on six years of record. Actual precipitation in the area is approximately 30 inches.

Given this prior information about the landfill setting, a study approach was chosen which would eliminate the necessity for an expensive borehole drilling program specifically to define the hydrogeology prior to initiating another drilling program for the establishment of suitable monitoring points. Instead, it was decided to use geophysical resistivity techniques, using a wenner configuration with electrode spacing of 65 feet, to map the areal extent of contaminant migration, as well as to gain an appreciation of the relative severity (concentration) of the contamination. Although the resistivity survey was not altogether successful, it did provide a suitable indication of the size of the conductivity plume. A detailed three-dimensional plume configuration was then obtained by conductivity profiling using a method of continuous augering and sampling. This technique employs a protected porous brass tip secured to a hollow stem auger. Water samples from any depth are withdrawn under suction through tubing attached to the brass tip and drawn through the augers. Samples of water were taken at 5-foot intervals in each location within the plume and field conductivity tests performed continuously on each sample. The data was used to plot the vertical distribution of the specific conductance. By profiling conductivities at several locations (cross-sections across the areal extent of the contaminant plume) the 3-dimensional extent of the contaminants in the subsurface was mapped.

The results to date show that the thickness of the sand varies from approximately 20 ft. to more than 80 ft., with the depth to the water table ranging from approximately 3 ft. to 33 ft. below ground surface. Based upon the specific conductance measurements, groundwater contamination has been restricted to the sand hydrostratigraphic unit and contaminants appear to have migrated a distance of approximately 2000 ft. from the landfill in the direction of groundwater flow. Contaminants are within the sand hydrostatigraphic unit moving over top of the clay soils within the sand hydrostatigraphic unit, with the main flux of contaminants being about 40 feet below the ground surface. Of particular significance with respect to this project is the uniformity in trends observed in the distribution of specific conductance values. The highest values were found near the landfill and they decreased consistently with distance from the landfill. In addition, in vertical profiles the highest readings were found near the centre of the contaminant plume with consistent decreases with distance above and below the centre of the plume. Thus a well-defined and geometrically simple contaminant plume appears to exist at the site. The simplicity and regularity of the plume is strong evidence of a simple hydrogeologic setting. Although this simplicity may not be typical of landfill environments, it does provide an ideal setting for the field study of leachate attenuation processes and the testing of numerical simulations procedures for predicting the migration of contaminants.

In order to establish a water budget for the site, rain gauge and evaporation pan equipment, water level recorders, infiltration lysimeters, snow level, and snow melt equipment was installed at the landfill. The only water quality parameter measured up to this time is specific conductance. Work is currently proceeding to complete the description of the hydrogeologic setting and to install a detailed network of piezometers for monitoring of the hydrogeologic conditions and groundwater chemistry. Long term monitoring of the detailed network will be undertaken to determine contaminant levels of certain selected parameters and to provide data for the mapping of specific contaminant enclaves.

Modelling Competition

Upon completion of the above field work at C.F.B. Camp Borden, it is anticipated that the results of the study can be used to evaluate predictive capabilities of ground water contaminant migration models as applied at landfills. Based on this evaluation, the usefulness of such models within a regulatory program can be assessed.

The state-of-the-art of groundwater contaminant migration modelling has progressed to the point where suitable models have been developed for predicting the behavior of landfill leachate contaminants within a well-defined hydrostatigraphic unit including both unsaturated and saturated zones. Most of these models are based on a general solute transport equation which can be stated as: (5)

$$\frac{\partial c}{\partial t} = \frac{\partial}{\partial x}\left(D\frac{\partial c}{\partial x}\right) - V\frac{\partial c}{\partial x} + \frac{1}{n}\frac{\partial q}{\partial t}$$

in which the first term on the right-hand side of the equation describes the movement of the solute due to dispersive effects, the next term describes the movement due to convective or bulk flow effects, and the last term accounts for contaminant attenuation. The variability in model capacity is attributed to both the manner in which the equation is solved and the make-up of the above attenuation term or its equivalent.

Other models are based on simplified physical-conceptual representations of the actual flow system, such as a chemical mass balance model simulating a series of completely mixed reactors and incorporating various sorption phenomena.

The desirability of utilizing some model for predicting contaminant migration, prior to the permitting of waste discharges to land, has been recognized by regulatory agencies. Some agencies have attempted, unsuccessfully to date, to develop "black-box" models for use in their regulatory permitting system, while others apparently have simply requested that a permit applicant be responsible for modelling his proposed discharge system for their adjudication. At a time when models are untested, and many uncertainties surround the use of such predictive "tools", the above practices are questionable.

If models have a place within a regulatory program, they must be tested for their predictive capabilities given a realistic amount of information which any permit applicant may reasonably be expected to provide. If the models require an excessive amount of detail in-put, or if they are incapable of providing answers without first having to be calibrated to a particular situation through the undertaking of a complete hydrogeological site investigation, or if they are prohibitively expensive to apply routinely, as would be necessary within any regulatory program, then clearly the models have proven to be useless, and any further efforts at progressing the state-of-the-art must be reassessed.

All modelling groups who have an interest in numerical simulation of landfill leachate contaminant plumes are invited to take the basic information to be contained in the CFB Camp Borden landfill study report, and to calibrate their model to simulate current conditions. Due to the absence of an accurate in-put function, detailing characteristics and quantities of waste components in the landfill and their historical rate of leaching and leachate characteristics it is necessary to define the present as the null condition. Using the study data as in-put, all participating modellers will be asked to provide a series of predicted contaminant migration patterns for the site, which will be checked against monitoring data to be routinely collected. Other than specific conductance and chlorides, a parameter list for monitoring has not yet been developed.

Furthermore, the same modelling groups may be provided information about a new CFB Borden landfill, which has been instrumented sufficiently to allow continued monitoring, such that the same calibrated model can be used to develop predicted migration patterns for this site in a somewhat more complicated hydrogeologic setting.

The proposed duration for this evaluation project is three years. At the completion of this period, predicted patterns will be compared with actual patterns. Continued monitoring will allow further comparisons in the future.

Peat Bog Landfilling

A study is being carried out on a landfill located in a peat bog type of hydrogeological setting, to determine the sub-surface movement of leachates and the potential for contamination of both the underlying aquifer and surface water.

The landfill under investigation is located in the Greater Vancouver area in British Columbia. The 365-acre site is the point of discharge for municipal, commercial, demolition, and other industrial and hazardous wastes. It is situated within a corner of one of the major peat bogs in the lower mainland area of British Columbia, and is probably a typical landfill environment on deltaic deposits in that region.

It was decided to carry out a comprehensive evaluation involving the investigation of the sub-surface geology by collating past bore hole information and the installation pf piezometer and monitoring wells for hydrogeological and sub-surface water chemistry ingestigations. Additional and confirming geological information was obtained in conjunction with the well installations.

From previous geologic investigations on the Richmond Landfill site, there was considerable bore hole data available. This data was plotted to show the site stratigraphy. The preliminary geologic setting developed indicated the site was situated on peat and overlying deltaic deposits of clay, silts and sands. The most interesting aspect of the geologic setting was its undulating stratigraphic units. Peats ranged in thickness from 5 to 20 feet.

Previous theories of leachate movement had assumed a continuous stratigraphy, with surface ditch interception of the total leachate flow. However, the geologic setting, determined from the preliminary evaluation of the geologic data, indicated that there were significant statigraphic highs and lows. This, coupled with the known compressibility of peat resulting from the landfill loading, indicated that interception of all the leachate by surface ditches may not occur. Hence, a thorough understanding of the hydrogeology was required.

Based on the known geology, a detailed drilling program was initiated to provide further borehole information, and to install monitoring wells for static/piezometric ground water level measurement and water quality monitoring.

The wells were located so as to provide hydrological and water chemistry monitoring at the base of the refuse, in the peats and below the clay. A background control well was located east of the landfill site. Piezometers were constructed of a 2 foot X 2 inch diameter plastic well screen, wrapped with fiberglass tape, attached to a 3-inch diameter plastic casing.

The monitoring activities on site included ground water sampling and analysis, drainage ditch water sampling and analysis, precipitation recording, and recording of tidal fluctuations of the adjacent Fraser River. Falling head or slug tests were carried out to determine the permeabilities of each hydrostatigraphic unit, including the refuse.

The results of the study are still being interpreted; however, it is evident that there are three distinctive statigraphic units, namely: the refuse which developed combined precipitation and tidal fluctuation response; the peat which had a piezometric level greater than that in the refuse and which was apparently responsive only to fluctuations of barometric pressure; and the underlying sand aquifer, which again showed tidal response. This information would tend to indicate an upward flow gradient in the peat unit; however, it is thought that the high piezometric levels in the peat are a result of peat consolidation and not of confined flow. Indications are also that the confining clay layer appears to retard leachate migration to the lower aquifer, although this is very much dependent on the substained integrity and continuity of the clay layer. There are indications that the clay layer is interspersed with sandy lenses which appear to be hydraulically connected to the lower aquifer. Some of these lenses may be exposed during the operation of the landfill.

Water quality evaluations indicate that leachate has not contaminated the underlying silty sand aquifer. Most leachate appears to flow towards and collect in the adjacent surface drainage ditches, which discharge directly to the Fraser River, although there is also significant flow away from the fill into unconsolidated peat layers. The unconsolidated peat, with a permeability of 10^{-3} cm/sec., is thought to have significant attenuation capabilities.

An interim report (6) has been prepared and is currently being revised and up-dated.

Leachate Recirculation

Another west coast landfill, situated on Vancouver Island north of the Greater Victoria area, is under investigation to determine the feasibility of leachate recirculation, over an extended period of time and under annual excess moisture conditions.

A study program was designed specifically to provide data on the effects of leachate recycle onto an existing major landfill. It will produce information which can be used at other sites as well as information specific to the test site. Information will be obtained in the following areas:

a) quantity of leachate emitted by the fill;
b) rate of fill stabilization;
c) disposal of water by evaporation and evapotranspiration; and,
d) use of the fill as a capacitance during peak flows of leachate.

It is recognized that a considerable amount of work (7,8) has been done elsewhere on the recycling of leachate which has shown that recycling under controlled conditions can accelerate stabilization of landfills and reduce contaminants in the leachate. It has also been found that heavy metals tend to build up in the leachate. The previous results are sufficiently encouraging to consider a full-scale field test on an active sanitary landfill. The study proposed extends the previous work to full-scale.

The Hartland Road landfill started receiving waste in 1959 and was operated for eleven years by open burning. Open burning was prohibited in 1970 and since that time the facility has been operated as a sanitary landfill.

The site is located in a bedrock saddle of igneous rock which forms a containment structure from which the only escape for surface water and leachate is a narrow channel. The landfill now covers about 40 acres, with a maximum waste depth of about 120 ft. The top of the fill is about 100 ft above the top of the fill.

The area receives about 40 inches per year of precipitation, primarily during the winter months, and this results in a leachate flow rate varying between virtually nil and 200,000 imp. g.p.d. with an average flow of about 500,000 imp. g.p.d.

In this study, which is just getting underway, it is proposed to recycle leachate by spray onto the surface of the completed portions of the fill to accomplish the following:

1. dispose of water by evaporation, evapotranspiration and absorption into the fill;
2. achieve treatment of the leachate on and in the fill, and,
3. accelerate stabilization of the fill area.

The study program is designed to establish relationships between spray application rate and retention capacity of the fill, evapotranspiration with selected ground cover, removal of BOD and COD, iron removal, tannin and phenol removal and the fate of heavy metals.

The program will continue over an initial three-year period assuming that equilibrium conditions are approached and that adequately representative data are obtained. The study may continue beyond that if fill equilibrium conditions cannot be established in this time.

The recirculation system and the landfill will be monitored by a variety of activities which include sampling the leachate discharge, sampling well points within the refuse, measuring fill settlement, and recording climatic data and fill temperatures.

The data to be obtained during the initial three year monitoring period will be used to determine the following:

a) Effect of recycle on leachate volume. Liquid input to the landfill will consist primarily of precipitation, recycled leachate and such intrusion water as escapes a surrounding interceptor system. The liquid output will consist of evaporation and evapotranspiration, and leachate discharge from the system. Ideally, this discharge to a stream is not a problem.

It is not always possible to match precipitation and rainfall and so the planned capacity of the fill for retaining liquid to compensate for inequalities in input and output becomes very important. The study will provide a water mass balance showing what performance may be expected under the test conditions. It is intended that a model will be prepared relating the input capacity and output under various exposure conditions.

b) Effect of recycle on leachate quality. The composition of leachate and the absolute amounts of each component will be investigated as a function of recycle rate. The principal changes in leachate composition should occur in the organic constituents. Inorganic components are expected to be less affected, although iron will be removed continually. As the interior of the fill changes from anaerobic to aerobic the organics should be destroyed more rapidly and the fill should stabilize more rapidly. The initial effect may be an increase in BOD, COD, and inorganics followed by a longer-term decrease in all parameters. It is the intent of this study to establish what course these parameters follow once recycling is instituted. Previous studies have shown a long-term decrease in BOD and COD with controlled water addition. The effect of recycling variable amounts of leachate, highly aerated, is not known. The data obtained will indicate what improvement in quality can be anticipated and what limitations may exist on recycle rate and leachate quality.

c) Effect of recycle on bacterial population.

d) Effect of recycle on soil cover vegetation. Within the scope of the study, the effect of recycling on the ability of grasses, shrubs and trees to survive and grow will be qualitatively evaluated.

e) The fate of heavy metals. It has been suggested that many heavy metals such as arsenic, lead, zinc, etc. build up in recycled leachate. If there is a net discharge of leachate, such build-up is undesirable. The extent and nature of such accumulations will be established.

f) The rate of stabilization of the fill as judged by gas evolution and composition, and settlement of the fill.

g) Effect of recycle on net volume of effluent discharged from the fill site.

The data which will be obtained will be applicable to any landfill at which leachate recycling may be either a feasible additional means of control of leachate discharges or a desirable alternative to treatment of leachate by chemical and biological means.

There is no data available at this time for further discussion.

SOIL-WASTE INTERACTIONS

Study Objectives

The main objective of our soil-waste interaction study program, involving several soil-waste projects, is to develop a means for predicting the attenuation and mobility of liquid or solubilized industrial wastes in soils, by investigating the soil-waste interactions in both saturated and unsaturated ground water regimes.

Whenever it is proposed to discharge industrial wastes to the land, it would be desirable to be able to predict the behaviour of the waste or its leachate within the soil system at the outset, and to predetermine the impact of the proposed discharge on the receiving environment.

At present, there is a need to study the mechanisms affecting waste attenuation and migration within the soil, and to develop quantitative information which can be used, directly or by extrapolation, to make the above assessments. When the data has been collected, it must be manipulated into a usable and workable form. This requires the development of a procedural format which can be used as a decision making "tool".

With this "tool", it should be possible for a regulatory agency to immediately assess any industrial waste disposal scheme involving a discharge to land, given the waste characteristics and the soil characteristics.

Literature Review Report

The first step towards this objective was to conduct a state-of-the-art-review, which was intended to provide background information for follow-up soil studies involving specific soil-waste interactions.

The specific objectives of the review study were to: review the literature on soil study techniques, waste attenuation mechanisms and factors affecting same, and contaminant migration as a result of the attenuating mechanisms involved; report on documented cases of ground water pollution resulting from industrial including hazardous and toxic waste discharges; and on the basis of the above, outline areas requiring further investigations.

In the resulting report (9), a review of the literature on the contamination potential of wastes in the soil-waste environment is presented. A conceptual model which integrates critical factors and processes affecting the attenuation, migration and fate of waste contaminants in soil is also discussed.

The report critically reviewed mathematical models which have been developed for the evaluation of the movement and fate of chemicals in the soil. The significance of particular soil study techniques is also discussed. Finally, documented cases of ground water pollution caused by discharge of specific industrial wastes are presented.

Soil-Waste Interaction Matrix

At the same time that the literature was being reviewed, a framework for the decision making "tool" was being developed, in the form of a generalized soil-waste interaction matrix which can be readily used as a simple means for assessing proposed industrial waste disposal sites. The matrix is to enable waste disposal recommendations to be made consistent with good land use practice and in accordance with acceptable environmental criteria.

The objective of the matrix development was to select a ranking scheme for wastes and a ranking scheme for soil-sites, and to combine these in a quantitative way in order to arrive at a disposal ranking for any waste soil-site combination. Achievement of this objective depended upon both selection and weighting of waste and soil-site parameters.

After the various waste characteristics and soil-site characteristics were defined, they were scored in points or arbitrary units. Each waste and each soil-site was then defined by a set of parameters. The waste parameters and the soil-site parameters become rows and columns in a matrix format. A description of the parameters and methods for calculating the various parameter values and for determining and utilizing matrix scores, as well as a complete description of the development of the matrix is contained in the final study report (10).

Both a site dependent and a site independent matrix were developed, the latter being a mini-matrix within the main matrix. The site independent matrix was intended to consist of rows of waste characteristics and columns of soil types. Basic attenuation information could be developed as in-put to the mini-matrix, and this information as a mini-matrix could then be used as direct input to a site-specific matrix, without having to constantly develop new site-specific information, even though experimental procedures for parameter quantification are defined in the report.

The matrix procedure is currently being tested, and at this time it appears to be a major contender in the ring of possibles for the U.S. EPA's attenuation procedure development work.

Batch Reactor Soil Study

As an alternate procedure for in-depth investigation of soil-waste interactions, and as a procedure which is capable of generating data for use in the above matrix procedure, the dispersed soil batch reactor technique developed at the University of Waterloo was tested with industrial waste. The test procedure used to study the attenuation and desorption of the waste liquids under anaerobic conditions, involved the sequential contacting of the soil and waste, followed by the desorbing water in a series of five dispersed soil reactors reflective of liquid movement through varying depths of soil. It is a deviation from the more traditional techniques involving soil column experimentation.

The laboratory procedure is described in detail in the final study report (11). With the resulting data, the mass of contaminant attenuated and desorbed was calculated by performing a mass balance calculation in each reactor for the contaminant examined.

The effectiveness of dispersion by soil water as an attenuation process was studied using chloride ion. It was possible to distinguish between attenuation due to dispersion and attenuation due to other mechanisms such as precipitation, mechanical filtration and sorption. It was shown that dispersion is a major mechanism of attenuation. The desorption of contaminants in the soil reactors was most effective for those contaminants attenuated primarily by dilution. For those contaminants attenuated primarily by mechanisms other than dispersion, desorption was found to be limited.

This method of study permits the evaluation of a zone of influence of a liquid waste disposal operation. Thereby guidance for the selection and operation of waste disposal sites is provided.

SPECIAL WASTE DISPOSAL

PCB Attenuation in Soils

A modification of the dispersed soil batch reactor procedure is being currently utilized in a study of the attenuation of Aroclor 1016 and Aroclor 1245 in soils. The soils vary in clay content, pH, cation exchange capacity, and organic matter content.

This study is designed to provide some basic information on the behavior of PCB's in soil under anaerobic conditions. It is known that a variety of PCB materials have been discarded at landfills across the country, and PCB's have been detected in landfill leachates in the parts per billion concentration range.

The study is scheduled for completion by June, 1977. Thereafter it is likely that a variety of other hazardous materials will be investigated in a similar fashion. The wastes of most concern at this time appear to be arsenic, mercury, phenols, and persistent organics.

Septic Tank Sludge Landfilling

The proper disposal of septic tank sludge also remains a problem. The raw sludge is generally not acceptable for discharge into a municipal sewer system, and simple sludge lagooning is also generally unsuitable in many locations. A significant quantity of this waste has been going into landfills, although recent concern about this practice on the effect of landfill leaching has caused many local authorities to disapprove of such practice.

The effect of septic tank sludge additions to municipal refuse was studied at the University of British Columbia, as part of a landfill leaching investigation conducted for the Province of British Columbia. It was shown in this preliminary work that the addition of septic tank sludge to mixed municipal refuse could significantly reduce the concentrations of contaminants in the leachate, under certain conditions(12).

If these preliminary findings could be confirmed, this would indicate that the discharge of raw sludge to landfills may be an environmentally sound practice. A follow-up study was initiated.

The objective of this study was to identify a range of application rates at which septic tank pumpings can be discharged to a landfill without increasing leachate contaminant loadings to a receiving environment. There appears to be a ratio of sludge weight to refuse weight above which an improvement of leachate quality will occur, and below which poorer leachate quality results. The determination of such a rate breakpoint would be required for design considerations.

The current lysimeter study is designed to investigate the behavior of varying sludge additions on leachate, under varying conditions of rainfall application and refuse depth.

Leachate is being analyzed for a minimum of 24 parameters, including total and faecal coliforms. The preliminary data appears to be confirming the previous findings that sludge additions can reduce peak contaminant concentrations by 40 to 100%. It appears that total masses of contaminants discharged are being reduced as well.

It was noted, however, that at the greater infiltration rates (45 inches per year) the peak concentrations are not reduced to the same extent. Therefore, increasing precipitation rates would appear to offset a fraction of these reductions. With only very early results available, it is premature to present a complete analysis at this time. The work is scheduled for completion this fall.

PROCEDURES DEVELOPMENT

Recommended Procedures Documents

The Waste Management Branch is currently reviewing methodologies for waste disposal site investigations and for site environmental protection systems design as part of our landfill research program. One of the objectives of this program is to develop a series of guideline documents on various recommended procedures, based on current technology and current thinking.

Landfill Leachate and Gas Monitoring

A series of international roundtable discussions was initiated to assist ourselves in documenting the current state-of-the-art, and in developing recommended procedures documents. To date, we have sponsored four separate sessions, each of which have been well attended by invited experts from government regulatory and research agencies, from consulting firms and universities, and from the waste disposal industry, in Canada, the U.S., and abroad.

The first session was held in April, 1975 in Vancouver, to deal with the topic of landfill leachate analysis. A report on recommended procedures for the analysis of landfill leachate, based on a consensus of opinion obtained from the delegates at the Vancouver session, was issued in October, 1975 (13).

A second session, dealing with ground water and soil sampling procedures was held in Ottawa in September, 1975. The session was held to exchange views and information on sampling techniques, and to obtain a consensus of opinion on methodologies which should be applied to the conduct of future landfill leachate migration studies, ground water quality monitoring programs, or any other investigations concerning the environmental impact of landfill leachate on ground water. A June, 1976 report presents recommended procedures based on the session proceedings (14).

Two other sessions have been recently held, one in Toronto to deal with procedures for the design and implementation of landfill monitoring programs, and the last one in Montreal which dealt with landfill gas detection, sampling, control, and extraction methodologies. A draft report based on the monitoring seminar is currently under review, and a report on the proceedings of the gas seminar should be received soon. Each of these reports will be used in the preparation of additional recommended procedures documents.

Leachate Toxicity Measurement

A specific procedure being developed is the measurement of leachate toxicity. Toxicity has often been suggested as a valuable indicator parameter in any monitoring of surface waters receiving landfill leachate discharges. However, standard bioassay procedures are expensive and generally require large volumes of effluent, and may take up to three days per sample. A rapid toxicity procedure has been developed to overcome these problems, and it has been demonstrated to be a less expensive, valuable field monitoring procedure.

The procedure, which is called the residual oxygen bioassay or the rapid toxicity assessment, was developed and tested in British Columbia at the Richmond peat bog landfill, utilizing a mobile lab unit inside a van.

All bioassays were conducted with 1.0 - 2.0 g. rainbow trout (Salmo gairdneri Richardson). Leachates were adjusted to pH 7 as required with HCl or NaOH prior to introducing test fish. The temperature maintained during bioassay was that of the holding tanks. Samples were aerated prior to bioassay to ensure that initial dissolved oxygen levels were greater than 90% saturation. Fish were not fed for 48 hours prior to starting the bioassays nor for the duration of the test. One or more control vessels of dechlorinated tap water was run for each bioassay.

Residual oxygen bioassays were carried out using serial dilutions of effluent in BOD bottles (300 ml). A selected number of test fish were placed in each bottle to give a loading density of 4.5 to 6.5 grams per liter. The bottles were stoppered with a water seal and incubated until all the fish in a test bottle had expired. Each bottle was removed from the bath after complete mortality and the dissolved oxygen was measured using a YSI model 54 oxygen probe or Delta Oxygen probe with stirrer.

Residual dissolved oxygen levels were plotted against effluent concentration on double logarithmic paper. By resolving the data on log-log scale into two groupings, a best fit straight line can be plotted for each group. One line has a zero slope,the "no-effect" or control line, and reflects the unimpaired ability of test fish to consume dissolved oxygen to control values. The second line, the "effect" line, has a positive slope where the toxicant had a limiting effect on oxygen utilization by the fish. The toxicant or effluent concentration at the intersection of these two lines is defined as the Threshold Limit Value (TLV).

Further details on the use of the procedure to evaluate toxic or non-toxic conditions in the field are provided in the final study report (15). The threshold limit values obtained from the rapid procedure were directly comparable to static 96-hour LC50 bioassays obtained by standard methods.

Municipal Waste Disposal Site Selection

Our other procedure development work includes the generation of a standardized simple, yet comprehensive, municipal waste disposal site selection procedure. The development of this systematic site selection procedure, which is intended primarily to assist small municipalities, is based on our evaluation of currently existing procedures, and also uses a matrix format. At this time it is anticipated that the procedure will involve an environmental protection matrix, a public protection matrix, an operational and management matrix, and a final decision matrix. The procedure will identify a sequence for matrix evaluation and site suitability scoring.

A final draft report on the procedure should be received by the end of March, 1977.

Waste Disposal Guidelines

Our last effort to be briefly mentioned in this paper relates to developing general guidelines for the disposal of various wastes onto land. In addition to preparing guidelines for specific wastes, such as wood wastes, septic tank sludge, etc., we are currently reviewing policy options in preparation for the development of overall guidelines on the discharge of municipal and industrial wastes.

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THE EFFECTS OF INDUSTRIAL SLUDGES ON LANDFILL LEACHATES AND GAS

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ABSTRACT

In an effort to assess the impact of the practice of co-disposal of industrial waste materials with municipal solid waste, a project utilizing large scale experimental landfill test cells was undertaken. Concern has been voiced that the addition of industrial wastes may result in the occurrence of various toxic elements in leachates and thereby pose a potential threat to potable ground water supplies.

The combination of municipal solid waste and various solid and semi-solid industrial residuals was added to several field lysimeters. All material flows were accurately measured and characterized for the continuing study. Data are presented on the mass flows for various chemical parameters within the generated leachates. Statistical evaluations were performed and the resulting data are discussed. Also, included are data on gas production and composition, as well as microbial activity.

INTRODUCTION

The disposal of industrial waste materials has been a matter of concern for some years now. All industrial waste residuals are not hazardous wastes, and, therefore some may be amenable to co-disposal with municipal solid waste. One aspect of this project has been to determine if certain selected industrial residuals were suitable for disposal in municipal landfills. These wastes were evaluated with respect to their effect on the decomposition of municipal solid waste, their effect on leachate and gas quantities and compositions, and their effect on microbial activity.

APPROACH

In an effort to evaluate the effects of the co-disposal of selected industrial waste materials with municipal solid wastes, large scale experimental landfills, were monitored.

The test cells (experimental landfills), employed for this study were epoxy coated steel, 1.8m (6ft) in diameter and 3.6m (12ft) in height; capable of holding approximately 3000kg (6600lbs) of municipal solid waste in a manner comparable to large area landfills. The size of the test cells was selected to minimize the problems of scaling factors generally associated with smaller laboratory lysimeters and to avoid the use of shredded refuse. Five (5) of the industrial waste test cells (cells 9, 10, 12, 13 and 14) were located outside and one of the industrial waste test cells (cell 17) was located in an enclosed bay area where higher ambient temperatures were maintained.

Prior to placement of any solid and/or industrial wastes, a layer of silica gravel, 300m deep, was placed in all cells as a base for the solid waste and to allow leachate to permeate to the drain system. All test cells were coated with coal tar based epoxy paint which was proven to be resistant to leachate. All test cells were loaded simultaneously in a period of five days.

Municipal solid waste from the City of Cincinnati was obtained directly from the packer truck for use in charging the test cells. Solid waste was placed into the test cells in 363kg (800lb) increments. The industrial residuals being evaluated were placed in the test cells simultaneously with the solid waste in every lift except the first. After each increment of waste was added, the solid waste or the solid waste/industrial waste mixture was mixed manually. Each lift was compacted to a height of 300mm (1ft) and a density of 470kg per cubic meter (800lb per cubic yard). This loading sequence was repeated eight times for each test cell to provide 2.4m (8ft) of compacted solid waste. The fully loaded test cells were then covered with 300mm (1ft) of compacted clay and all cells were sheltered from both moisture and sunlight. Temperature monitors were installed throughout all cells and in the soil at various locations. Water addition to the cells is at a rate of 406mm (16in) per year and is accomplished on a monthly basis in accordance with anticipated net infiltration for the midwestern portion of the country.

INDUSTRIAL RESIDUALS BEING EVALUATED

The industrial process residuals studied included a refinery sludge (RS), a battery production waste (BPW), an electroplating waste (EW), an inorganic pigment waste (IPW), a chlorine production brine sludge (CPBS), and a solvent based paint sludge (SBPS). Physical characteristics and amounts of industrial wastes added to each of the test cells are provided in Table 1.

Notable components of the sludges were: copper, iron, mercury and moisture for the refinery sludge; copper, iron, cadmium, lead, asbestos, tin, antimony and moisture for the battery production waste; chromium, iron, arsenic, cadmium, cyanide and moisture for the electroplating waste; beryllium, chloride, asbestos, clay volatile fibers and moisture for the inorganic pigment waste; and nickel, lead, chloride, asbestos, mercury and clay volatile fibers for the chlorine production brine sludge. A complete chemical analyses of these waste materials is shown in Table 2.

SAMPLE COLLECTION AND ANALYTICAL METHODOLOGY

Accurate and precise determinations of chemical parameters on landfill leachates have always been difficult. In an effort to retain anaerobic conditions, all leachates were collected under an argon atmosphere. With only several exceptions, the analytical methods as described by Chian and DeWalle[1], and those recommended by Environment Canada[2], were followed. Several modifications to these methods were employed but will be discussed in more detail by Mr. Richard Carnes. Precision and accuracy were determined for many of the chemical parameters being evaluated. With the exception of oxidation-reduction potential and volatile organic acids the analytical determinations proved quite precise and accurate.

Accurate determination of gas quantity and composition proved even more difficult than the chemical analyses. We had noticed gas permeation in several of the experimental landfills. This leakage was a result of several factors, the most significant of which was the permeability of the materials used in the construction of the test cells. A permeability study of the construction materials indicated that all polymeric materials used in the test cells were permeable to many of the gases being produced at the flow rates we were experiencing.

TABLE 1. INDUSTRIAL WASTE PHYSICAL CHARACTERISTICS

Cell	Waste Type	Moisture[a] Content	Amount Added[b]	Characteristics
9	RS[c]	79.00	1518	High bacterial activity, black moist
10	BPW[d]	89.25	1291	Grey-lg amount of liquid
12	EW[e]	79.53	1191	Brown, soupy
13	IPW[f]	51.75	1420	Black, solid, no odor
14	CPBS[g]	24.11	2039	Very dense, no odor, light brown, moist
17	SBPS[h]	24.75[i]	1604	Red to white color, putty consistency, strong odor

[a] Percent by wet weight.

[b] kg.

[c] Refinery Sludge.

[d] Battery Production Waste.

[e] Electroplating Waste.

[f] Inorganic Pigment Waste.

[g] Chlorine Production Brine Sludge.

[h] Solvent Based Paint Sludge.

[i] Mainly organic solvents.

TABLE 2. INDUSTRIAL WASTE CHEMICAL ANALYSIS[a]

Waste Cell Number	RS[b] 9	BPW[c] 10	EW[d] 12	IPW[e] 13	CPBS[f] 14	SBPS[g] 17
Total Solids[h]	21.00	10.75	20.47	48.25	75.89	75.25
Total Volatile Solids[h]	31.00	7.94	8.98	22.25	1.17	55.31
Moisture[h]	79.00	89.25	79.53	51.75	24.11	24.75
Cr	125	155	1.56[h]	0.50	5.00	75.0
Ni	23	32	35	10	65	0.5
Cu	3500[i]	1125	100	110	125	2.0
Fe	5560	2950	1.37[h]	1000	2000	150
As	1.0	72	460	3.4	14.5	12.8
Be	4.8	1.8	0.25	20.2	<1.0	<1.0
Se	26.0	180	4.50	16.0	16.5	7.60
Cd	0.50	29.0	38.5	10.5	0.70	0.50
Cn	1.0	4.2	460	3.4	14.5	12.8
Pb	182	3.48[h]	267	120	697	12.6
Cl[h]	2.35	1.12	1.35	10.0	20.0	0.75
Asbestos[j]	3.00	208	23.0	45.0	110	9.00
Hg	10.6	4.80	14.7	7.60	227	16.7
Sn	NA[k]	6800	NA	NA	NA	NA
Sb	NA	1.32[h]	NA	NA	NA	NA
Clay Volatile Fibers[j]	40.0	720	86.0	185	480	65.0
Zn						
V	NA	120	NA	40	NA	NA
B	7.20	8.10	19.0	28.5	1.70	11.4
Ti	NA	NA	NA	NA	<0.1	NA

[a] All values in ppm unless otherwise specified.

[b] Refinery Sludge.

[c] Battery Production Waste.

[d] Electroplating Waste.

[e] Inorganic Pigment Waste.

[f] Chlorine Production Brine Sludge.

[g] Solvent Based Paint Sludge.

[h] Percent by wet weight.

[i] Underlined values indicate maximum sludge concentrations.

[j] Fibers/100 g.

[k] Not analyzed.

Since the test cells were placed outside and below the ground level, it was impossible to correct many of those problems. The experimental landfills placed in the bay area however, have been sealed and volumetric collection of gases has proceeded. The ability of landfill gases to permeate polymeric materials is significant and should be considered a factor in the assessment of synthetic liners for actual land disposal operations.

Determination of gas composition was accomplished employing gas chromatography with thermal conductivity detectors. Analysis of gas samples should begin immediately after collection since error is introduced when samples are stored in the collection vessels. This is due to the permeability of the access septum to various gaseous constituents.

LEACHATE COMPOSITION

Since the composition of leachate emanating from the solid waste only and solid waste/industrial waste test cells has been discussed in previous papers[3,4] it will only be briefly described here. Chemical analysis of leachates from the solid waste only test cells and those containing industrial wastes admixed with municipal solid waste shows very little variation for organic, nutrient and demand parameters. Figure 1 indicates the typical differences encountered in the total organic carbon removals between the municipal solid waste only test cell and those containing municipal and industrial waste mixtures. The exception to this is the refinery wastes, which has shown considerably lower values for both total organic carbon and chemical oxygen demand, Figure 2. This ambiguity in the data will be discussed in more detail in a subsequent section.

Figure 2 shows typical mass flows for chemical oxygen demand (COD). Once again, similar results are obtained for the municipal solid waste only test cells and those containing the industrial waste additions. Graphical displays of total Kjeldahl nitrogen (TKN) are shown in Figure 3. Again, the industrial waste/solid waste test cells behave in a comparable manner to the solid waste only test cells.

One point which must be made at this time is that the solid waste only test cell has received twice the amount of water that the industrial waste test cells have. This comparison is necessary because of the high initial moisture content present in the industrial wastes which has effectively reduced the amount of infiltrating water required to reach field capacity.

Graphical displays for various metallic ions are given in Figures 4, 5 and 6. Analysis of the leachates for many of the metallic ions present in the industrial wastes has indicated considerable variation between the test cells. This variability in leachate composition is relatable to the initial chemical composition of the industrial waste itself. For example, the lead content of the battery production waste was considerably higher than that for the municipal solid waste. We would, therefore, expect to see higher concentrations of lead within the leachate of this test cell. These assumptions have been borne out.

Analysis of the industrial waste/solid waste leachates for highly toxic components, such as, hexavalent chromium and methyl mercury, indicates slightly higher concentrations present in some of the leachates. These concentrations however, are in the low ppb range and we have not been able to successfully evaluate their statistical significance.

STATISTICAL EVALUATION

Correlation coefficients have been determined for a number of analytical parameters. Parameter concentrations were normalized by leachate volume and then compared from cell

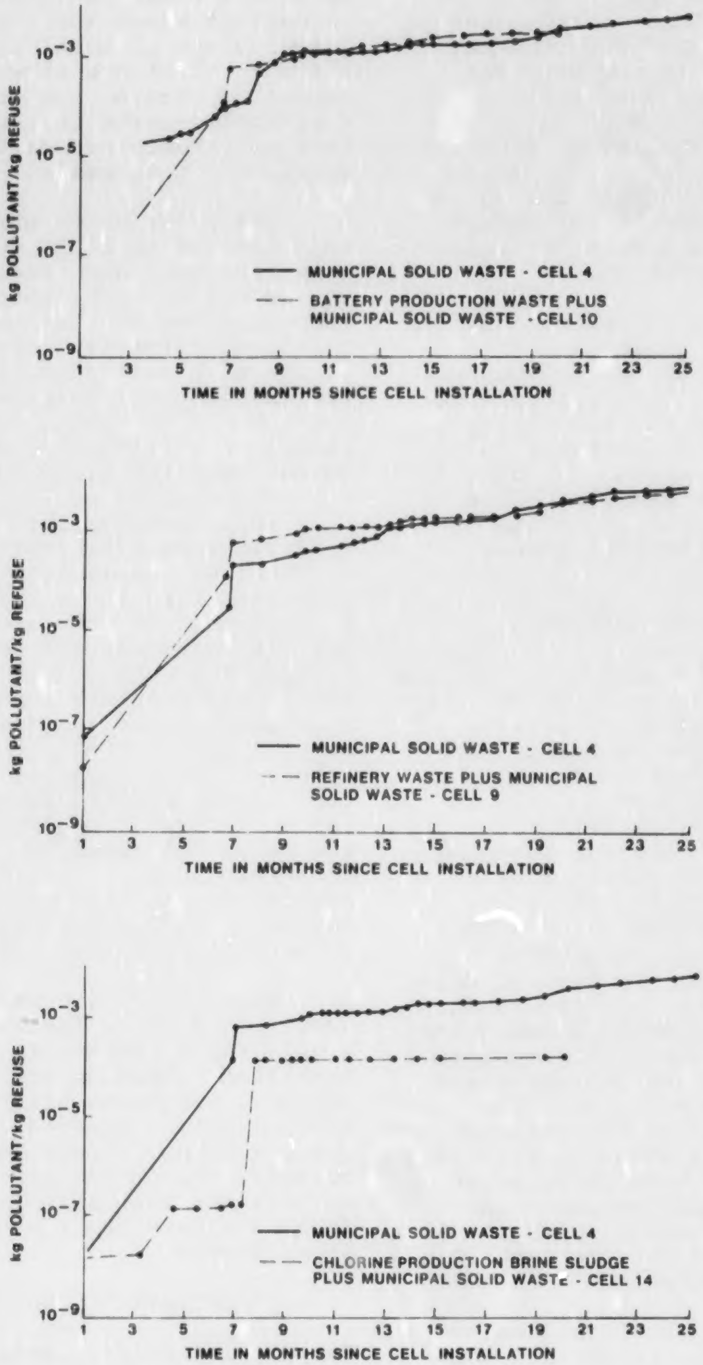

FIGURE 1. TOC REMOVED THROUGH LEACHING

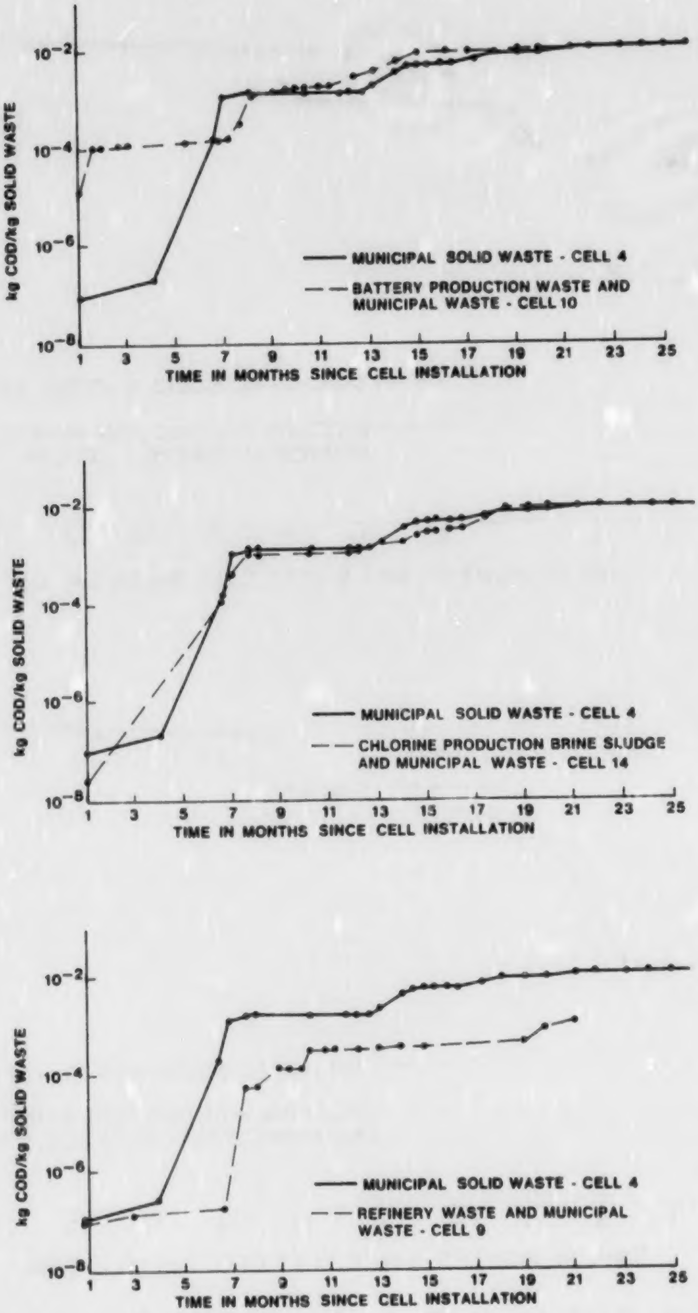

FIGURE 2. COD REMOVED THROUGH LEACHING

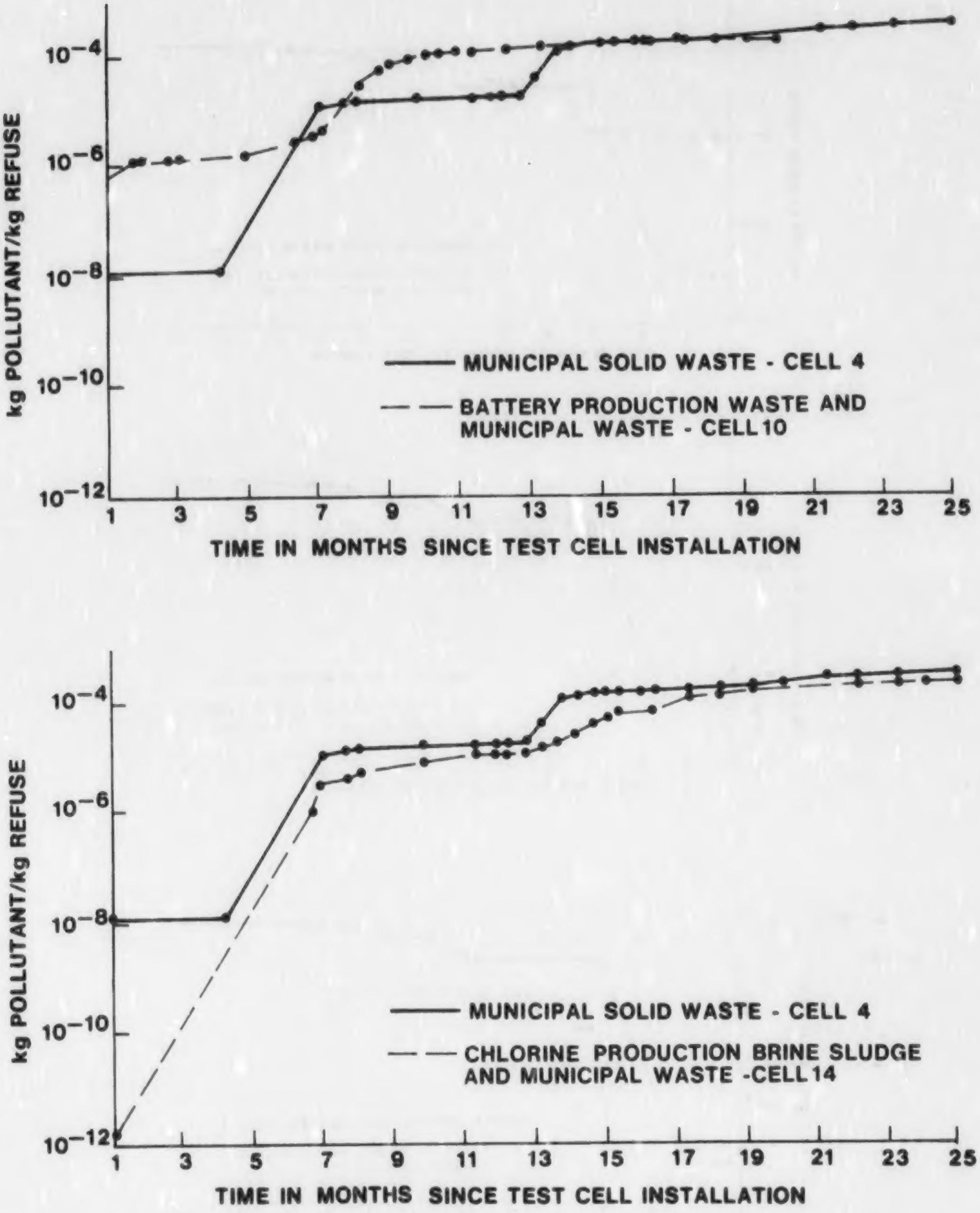

FIGURE 3. TOTAL KJELDAHL NITROGEN REMOVED THROUGHLEACHING

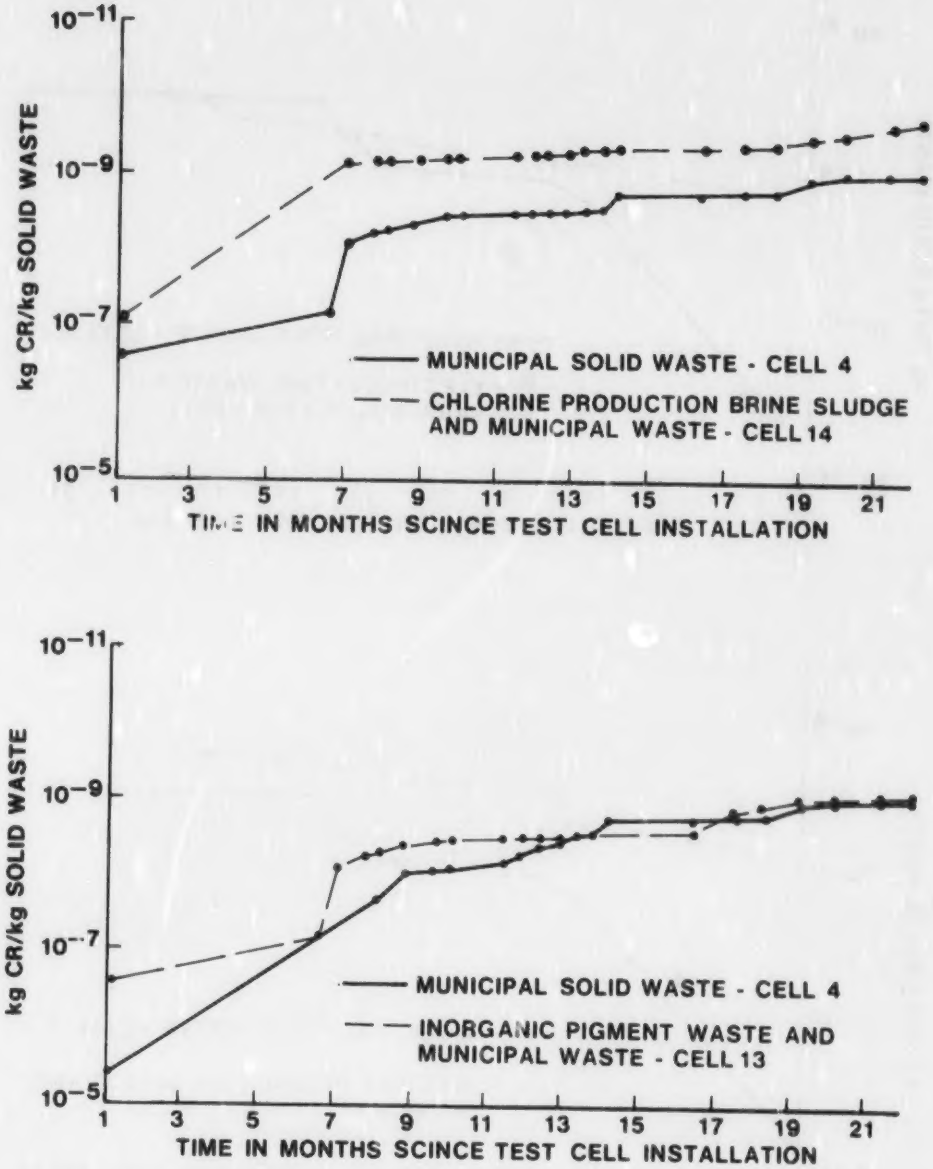

FIGURE 4. CHROMIUM REMOVED THROUGH LEACHING

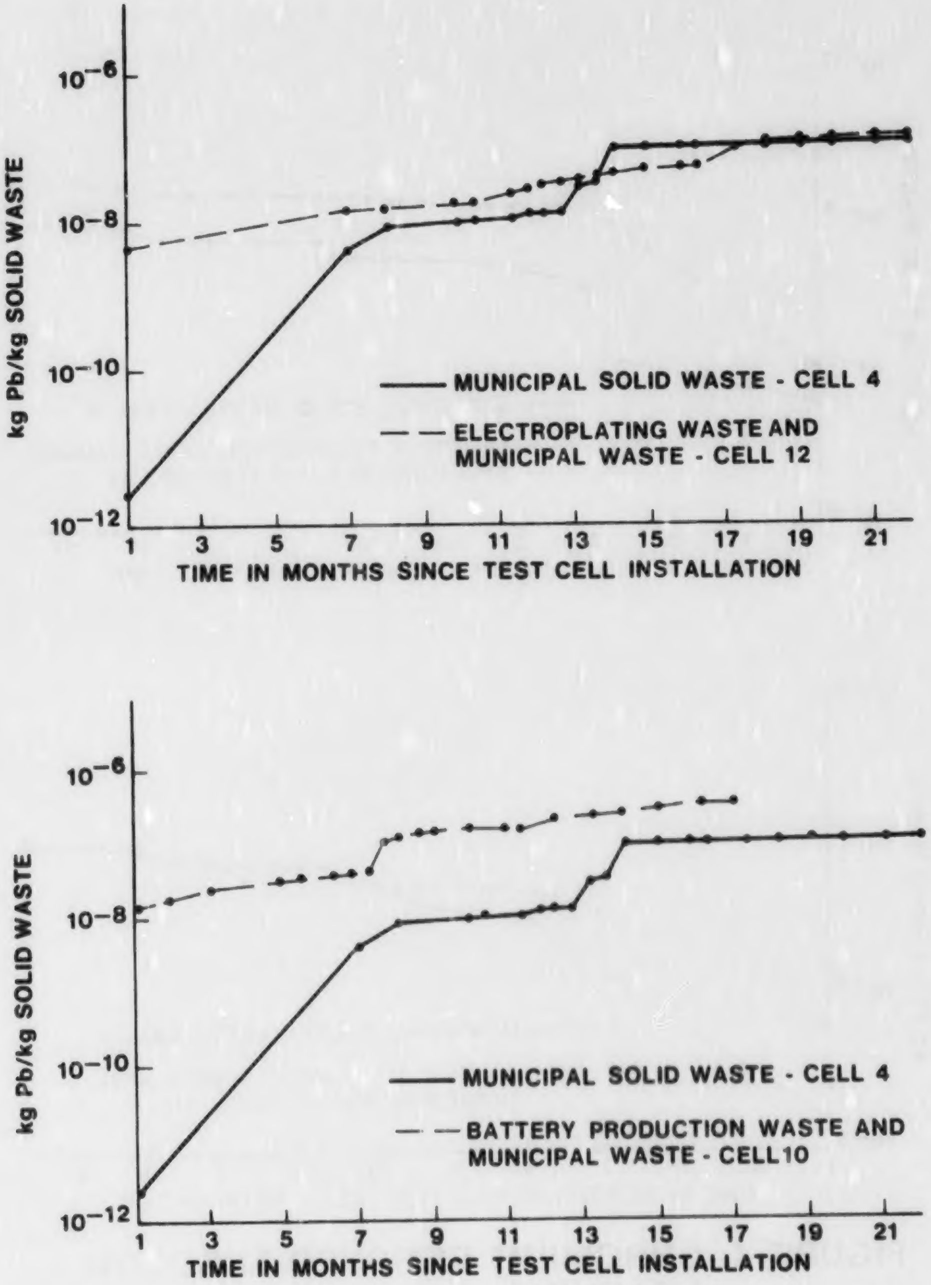

FIGURE 5. LEAD REMOVED THROUGH LEACHING

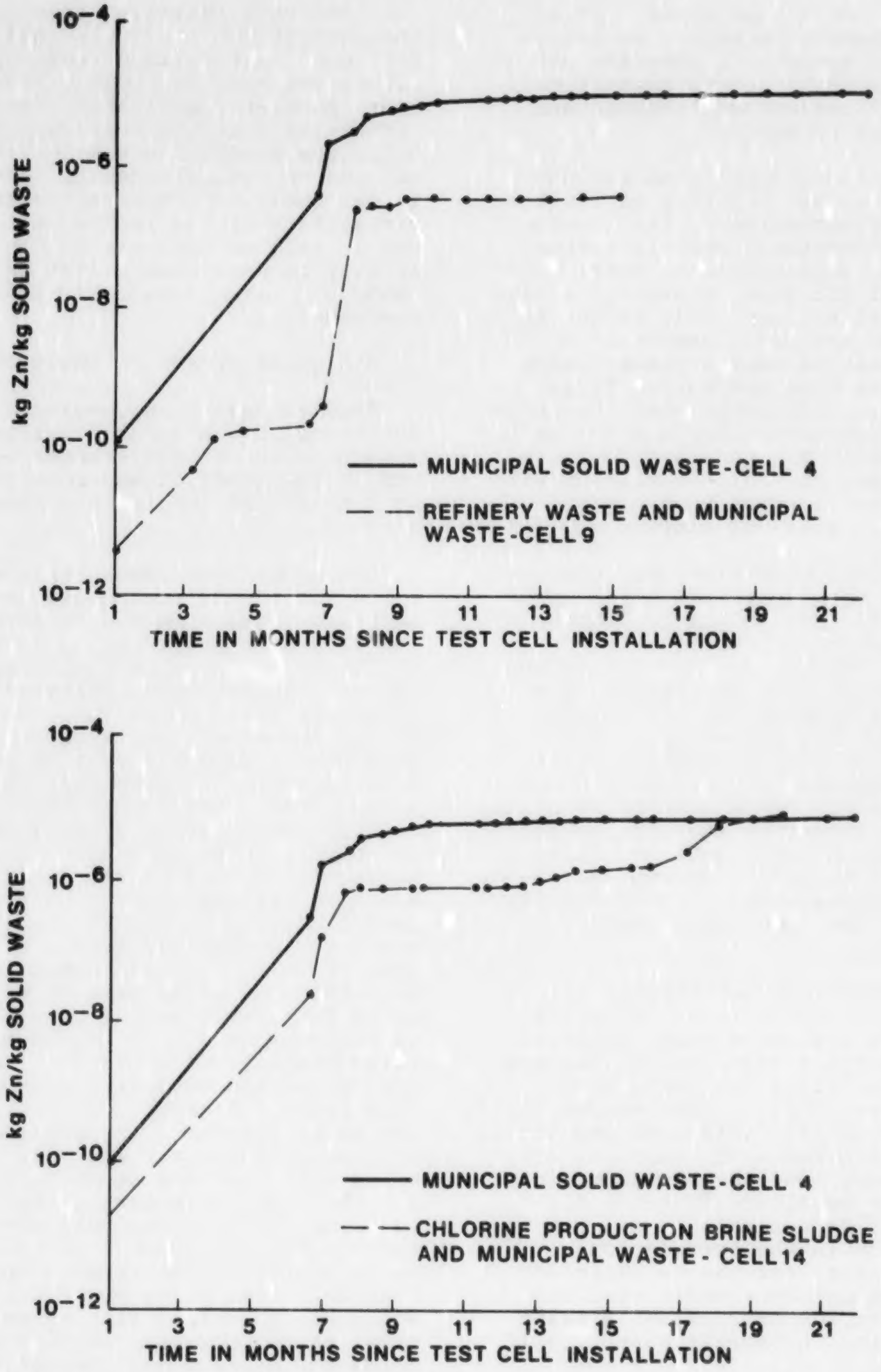

FIGURE 6. ZINC REMOVED THROUGH LEACHING

to cell on a time basis. Parameters were selected which represented a variety of possible pollutants, such as demand parameters, nutrient parameters, organic and inorganic parameters.

Excellent correlation was obtained for the majority of the test cells irrespective of the treatment applied for total organic carbon, chemical oxygen demand, total Kjeldahl nitrogen, dissolved solids and total solids. This is not to say that the total amount of a particular parameter leached from the cells does not vary. It is simply an indication that the shape of the concentration/time curves for these cells are very similar. In most cases the cells correlate with each other for the parameters mentioned at a 99% confidence limit.

As previously discussed the refinery waste/municipal solid waste test cell (cell 9) is generating a leachate whose constituents are present in very low concentrations. It also has shown extremely poor correlation with all other test cells for all parameters. This means that either degradation is occurring and the parameter concentrations are being diluted by infiltrating water, or that the degradation of the waste material is very slow due to toxic effects exerted on the microorganisms by some constituent of the industrial waste material.

Correlations for the metallic ions vary considerably depending upon the treatment under evaluation. All municipal solid waste only test cells correlate well with each other for metallic ions and hardness. The test cells containing industrial wastes admixed with municipal solid waste vary considerably depending upon the metallic ion being discussed. In general, it appears that variation in the correlation coefficients for the metallic ion is dependent once again upon the concentration of that ion present within the industrial waste itself.

One very positive aspect of the correlation study was that the replicate test cells correlated within 99% confidence limit for every parameter evaluated. This is indicative that the replicate test cells are generating similar physical and analytical results. This is extremely important since the variability of the replicate test cells relates directly to the ability to determine variation among all other treatments being evaluated.

GAS QUANTITY AND COMPOSITION

Studies have been completed to determine if gas stratification by molecular weight differences occurs within the landfill environment. No such stratification has been seen.

Gas production rates for the solid waste only test cells have averaged between 40 and 50 liters per day, or approximately, 13 to 15 milliliters of gas per kilogram of refuse. It has been estimated[5] that each kilogram of solid waste during decomposition is capable of producing 1.22 cubic feet of carbon dioxide and 1.77 cubic feet of methane for a total of 2.99 cubic feet per kilogram of solid waste. It is felt that conversion of between 25 to 75% of the solid waste to gas may occur within a landfill system[6]. Therefore, the total gas production from a kilogram of solid waste is estimated to be within the range of 0.77 to 2.24 cubic feet per kilogram of solid waste decomposed. If we assume the estimated life of a landfill to be ten years, theoretical gas production rates for these test cells can be calculated. Theoretical gas production rates for the test cells being evaluated are shown in Table 3. The test cells containing municipal solid waste only approach the maximum gas production that can be theoretically achieved. The test cell containing municipal solid waste admixed with an industrial waste, however, is slightly below the minimum gas production that would be expected under these

TABLE 3

THEORETICAL AND ACTUAL GAS PRODUCTION RATES

CELL	CONTENT	AMOUNT SOLID WASTE(Kg)	THEORETICAL GAS PRODUCTION Min(ft^3)	Max(ft^3)	ACTUAL GAS PRODUCTION ft^3
16	Solid Waste Only	2996	2307	6711	4134
17	Solid Waste and Industrial Waste	2998	2308	6715	2084
*18	Solid Waste Only	3000	2310	6720	6300
*19	Solid Waste Only	3012	2319	6747	6024

*These test cells were loaded at a later date and are in the non-methanogenic phase of decomposition during which time large quantities of carbon dioxide are produced. This accounts for the higher amount of gas production.

conditions. We can, therefore, assume that some adverse effect is being exerted by the industrial waste present within this test cell.

CONCLUSIONS

1. Existing analytical methods are available for the accurate and precise determination of chemical parameters within landfill leachates. Replicate analyses and standard addition samples should be analyzed also to insure validity of the results.
2. Gas permeation of polymeric and synthetic materials occurs in substantial quantities. This should be considered when evaluating liner materials for actual land disposal sites.
3. High moisture content industrial residuals with municipal solid waste appears to have no effect on organic, nutrient and demand parameters.
4. The addition of the evaluated industrial residuals with municipal solid waste appears to have no effect on organic, nutrient and demand parameters.
5. The co-disposal of industrial residuals with municipal solid waste may have an effect on the metallic ion concentration within the leachate. This is highly dependent upon the chemical composition of the industrial waste itself.
6. Gas stratification by molecular weight does not appear to occur within a landfill environment.
7. Certain industrial waste residuals may have a significant effect upon gas production rates within a landfill environment. The theoretical gas production rate of between 1.0 and 2.2 Cubic feet per kilogram of solid waste decomposed appears to be valid.

ACKNOWLEDGEMENTS

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INFLUENCE OF MUNICIPAL SOLID WASTE PROCESSING ON GAS AND LEACHATE GENERATION

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ABSTRACT

The purpose of the experiment is to evaluate the effect of differing preplacement processing techniques on the production of gas and leachate by landfilled municipal solid waste. Five in-ground, concrete test lysimeters, 7 ft. x 11 ft. x 12 ft., were prepared and filled with processed refuse and then sealed. The types of processed refuse studied are: shredded unbaled refuse (Cell 4); shredded baled refuse (Cell 1); baled whole refuse (Cell 2); and baled whole refuse placed in a "saturated" condition (Cell 3). One lysimeter contains unprocessed whole refuse as a control (Cell 5).

This paper presents the results of the first two years of observing the completed lysimeters. Leachate volume and composition and gas composition histories are discussed. Gas volume data is incomplete due to periodic failure of the gas seals on the lysimeters. Correlation coefficients were calculated for the pollutant production data to compare the relative effects of the five studied processing options. Such comparison shows that the time histories of pollutant production are similar in nearly every case. Differences in magnitude of total pollutant are too small to conclude that the cumulative emissions do not belong to the same set.

INTRODUCTION

The intent of the present program was to determine the effect of different pre-landfilling processing techniques on quantity and composition of gas and leachate produced from municipal solid waste in a landfill environment. The processing options of interest were shredding, shredding and baling, and baling alone. At the same time, it was of interest to determine the effect of a highly moist (saturated) environment on gas and leachate production and to compare all treatments to the emissions from unprocessed waste.

To accomplish this goal, a facility consisting of five identical test lysimeters and an instrumentation cell was designed. Each test lysimeters were designed to hold approximately 10,000 kilograms of municipal solid waste when completed. The cells were constructed inground and, except for the "saturated" test cell, were subjected to a moisture regimen simulating the rainfall pattern of the Midwestern United States. In order to compare the processing options, Cell 1 was loaded with baled shredded refuse; Cell 2 with baled, whole refuse; Cell 4 with unbaled, shredded refuse; and Cell 3 with baled, whole refuse subjected to a high moisture regimen. Unprocessed whole refuse was loaded into Cell 5 as a control for comparison purposes. These cells were constructed, loaded with municipal solid waste, and sealed. They have been continuously monitored since January, 1975.

FACILITY DESCRIPTION

The test cells were constructed of reinforced concrete and have inside dimensions of 2.1m by 3.4m by 3.7m. The walls are 20cm thick reinforced concrete, and there is a separation of 1.1m between each cell. Instrumentation access to the interior of the cells was provided by casting sleeves into the front cell wall and then installing bulkhead fittings.

A data collection cell is located in front of the middle test cell and is designed to contain the terminals and collection ports for all gas, leachate, temperature, and moisture monitoring equipment.

The interior of the test cells was intended to simulate a typical sanitary landfill, while still permitting the required sampling and monitoring. Therefore, each test cell contains a 14cm base of non-reactive silica gravel, three layers of baled refuse or 2.7m of compacted refuse, covered by 30cm of compacted clay with 30cm of pea gravel above that. Each test cell also has 15cm of free-board above the pea gravel. A water injection rake is buried in the pea gravel in an attempt to distribute the water uniformly over the top of the cell so that moisture might seep gradually into the waste in a uniform manner, similar to that which would occur when rain falls on a landfill.

The cells are instrumented for temperature and moisture measurements within and have gas and leachate collection lines emanating from the cells. Thermocouples are located at 24 points distributed throughout the refuse in each cell to determine the temperature profile within each cell.

The moisture monitoring equipment placed within the cells consisted of both gypsum soil blocks and porous cup tensiometers. Gypsum soil blocks are merely water absorptive resistors. As moisture is absorbed, their electrical resistance changes and is measured with an ohmmeter, calibrated to directly read moisture content. There are nine gypsum soil blocks in each test cell.

The porous cup tensiometer consists of a porous ceramic tube filled with water. The tendency of the water to flow through the ceramic tube will depend upon the partial pressure of water outside the tube. A vacuum gage connected to the ceramic tube allows measurement of the water remaining with the tube, and thus indirectly, of the moisture outside the tube.

The floor of each test cell slopes slightly toward the center front where a leachate collection line is located. These lines pass through to the instrumentation cell, where they are valved to permit periodic removal of leachate collected in the bottom of the cells.

The original gas monitoring system installed in the cells consisted of 10 collection probes, 9 of which were located in the solid waste and one of which was located in the head space of each cell. These collection lines passed through the front wall of the test cells and were manifolded and valved and connected to a precision wet test gas meter. The gas collection and monitoring system proved to be totally inadequate and was modified, as will be discussed in a later portion of this paper.

Before loading the test cells, all the interior surfaces were coated with coal tar epoxy to prevent leaching of the concrete cell walls. A cast concrete cell cover was placed on top of the cells after loading. Gasket material made of vinyl tubing was installed between the cell wall top and the cell covers in an attempt to assure gas-tight integrity. Figure 1 is an isometric showing the outside of a typical test cell used in this study.

SOLID WASTE LOADED

All solid waste had been planned to be obtained in the Dayton, Ohio area, the shredding and baling to be done in appropriate facilities near Dayton, Ohio. A fire at the originally proposed baling facility precluded this, and the only baling system available for use at the time was one located in Cobb County, Georgia. Fortunately, a shredder facility was also available in nearby DeKalb County. So, municipal solid waste was obtained in DeKalb County and shredded there, transported to the Cobb County baler facility, baled and transported to the test site at Franklin, Ohio.

Unbaled waste was obtained from Oakwood, Ohio. Some of that waste was shredded at the Montgomery County, Ohio shredder facility in the south incinerator plant prior to transport to the test sites at Franklin, Ohio. Samples of the different refuses were categorized using hand-sorting techniques to determine the composition of the loaded wastes. Results of this categorization are shown in Table 1.

The baled refuse was loaded into the cells and backfilled with clay in an attempt to retain the high compaction density in the bales (850 kilograms per cubic meter). Nonbaled refuse was placed and compacted to 550 kilograms per cubic meter for the shredded waste and 500 kilograms per cubic meter for the unshredded waste. All waste was covered with clay, which was then compacted. Finally, all were covered with pea gravel and sealed with precast concrete cell covers. The monitoring program then officially began.

MONITORING

As indicated earlier, temperature was measured within each test cell with an array of 24 thermocouples. Test cells uniformly show a history of following the ambient outside temperature, with the bottom portion of the cell being warmer in winter and cooler in summer than the top. The vertical temperature differences were in the range of 5°C to 10°C. The tops of the test cells experienced a seasonal temperature variation in the range of 40°C, while the bottoms of the test cells experienced only an 18°C seasonal temperature variation. Comparing these findings to the ambient ground temperature indicates that the test cell contents are a few degrees cooler than the ground in winter and a few degrees warmer than the ground temperature in summer. This indicates that the test cell construction materials are better heat conductors than the ground itself.

There were 11 moisture probes in each test cell. There has been a fairly high failure rate (about 40 percent) of the gypsum block moisture probes. The moisture probes indicated some initial short-circuiting, but now they indicate uniform moisture throughout the test cells. Gypsum block probes are presently reading 100 percent saturation within the test cells. This is not borne out by the moisture retained curves presented as Figures 2 through 5. This discrepancy may be due to the fact that moisture probes were calibrated for water while the leachate within the test cells has a significantly different conductivity than the

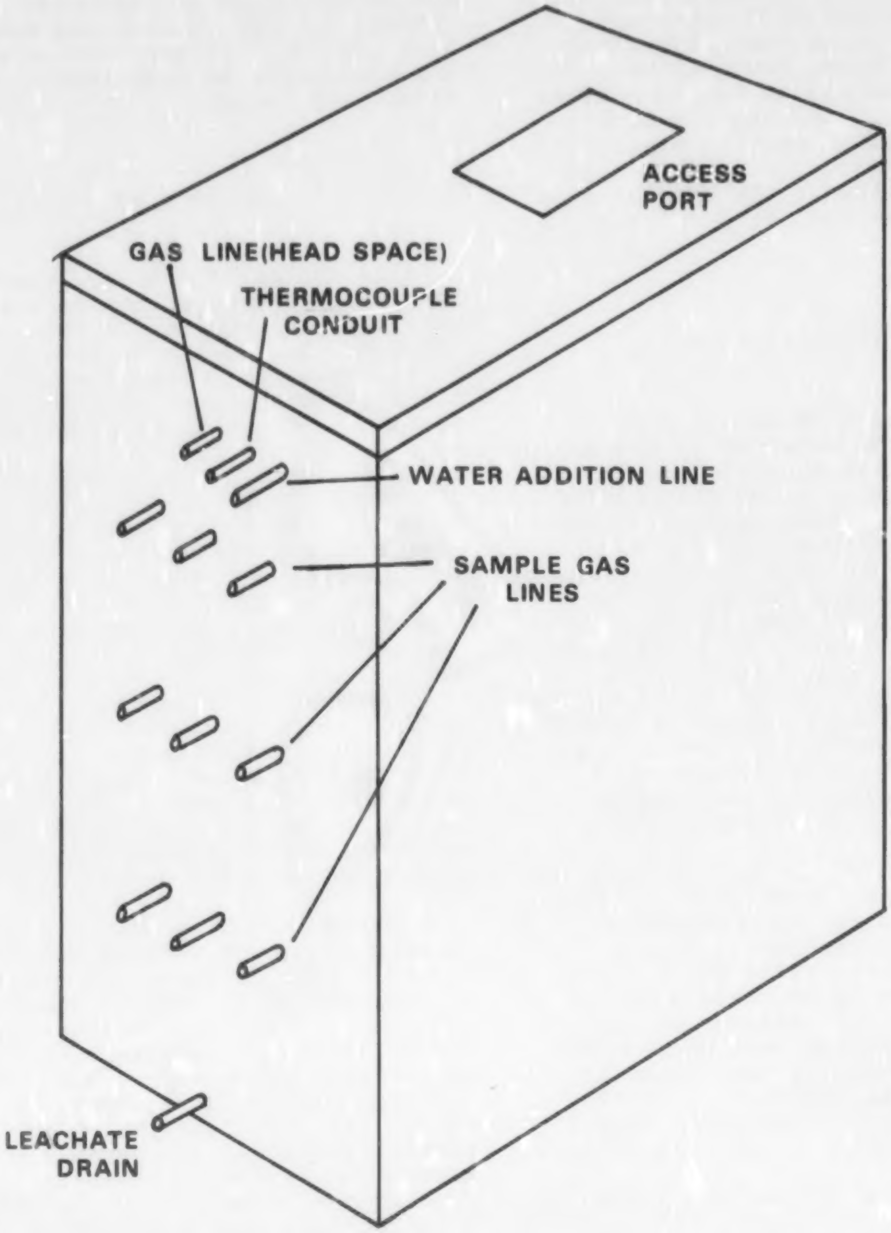
ACCESS PORT
GAS LINE(HEAD SPACE)
THERMOCOUPLE CONDUIT
WATER ADDITION LINE
SAMPLE GAS LINES
LEACHATE DRAIN

FIGURE 1. CONFIGURATION OF TEST CELLS

TABLE 1 REFUSE CATEGORIZATION

(PERCENT BY WEIGHT)

CATEGORIES									
CELL	Cell 1	Cell 2&3	Cell 2&3	Cell 4	Cell 4	Cell 5			
SOURCE CONDITION	Dekalb Shredded	Cobb Unshred.	Cobb Unshred.	Oakwood Shredd.	Oakwood Shredd.	Oakwood Unshred.	Oakwood Ave.	Atlanta Ave.	Ave.
FOOD WASTE	7.9	4.4	8.5	6.6	8.9	7.3	7.6	6.9	7.3
GARDEN WASTE	1.6	0.9	1.0	8.3	10.0	5.5	7.9	1.2	4.6
PAPER	45.2	48.8	53.2	57.0	44.5	48.6	50.0	49.1	49.6
PLASTICS, RUBBER LEATHER	6.6	9.4	6.1	3.7	4.4	5.5	4.5	7.4	6.0
TEXTILES	3.8	2.5	-	2.7	4.7	5.3	4.2	2.1	3.2
WOOD	0.6	0.6	0.5	1.2	1.2	2.3	1.6	0.6	1.1
METAL	7.9	12.0	15.0	5.8	8.9	7.3	7.3	11.6	9.5
GLASS	11.1	17.3	13.6	9.7	8.6	11.9	10.1	14.0	12.0
ASH, ROCKS DIRT	4.8	1.4	0.7	1.6	3.3	2.8	2.6	2.3	2.4
DIAPERS	5.4	0.9	-	0.8	0.5	0.7	0.7	2.1	1.4
FINES	4.8	1.6	1.4	2.5	5.0	2.8	3.4	2.6	3.0
MOISTURE % BY WT.	28.2	34.8	21.2	27.0	33.3	24.9	27.2	27.3	27.25

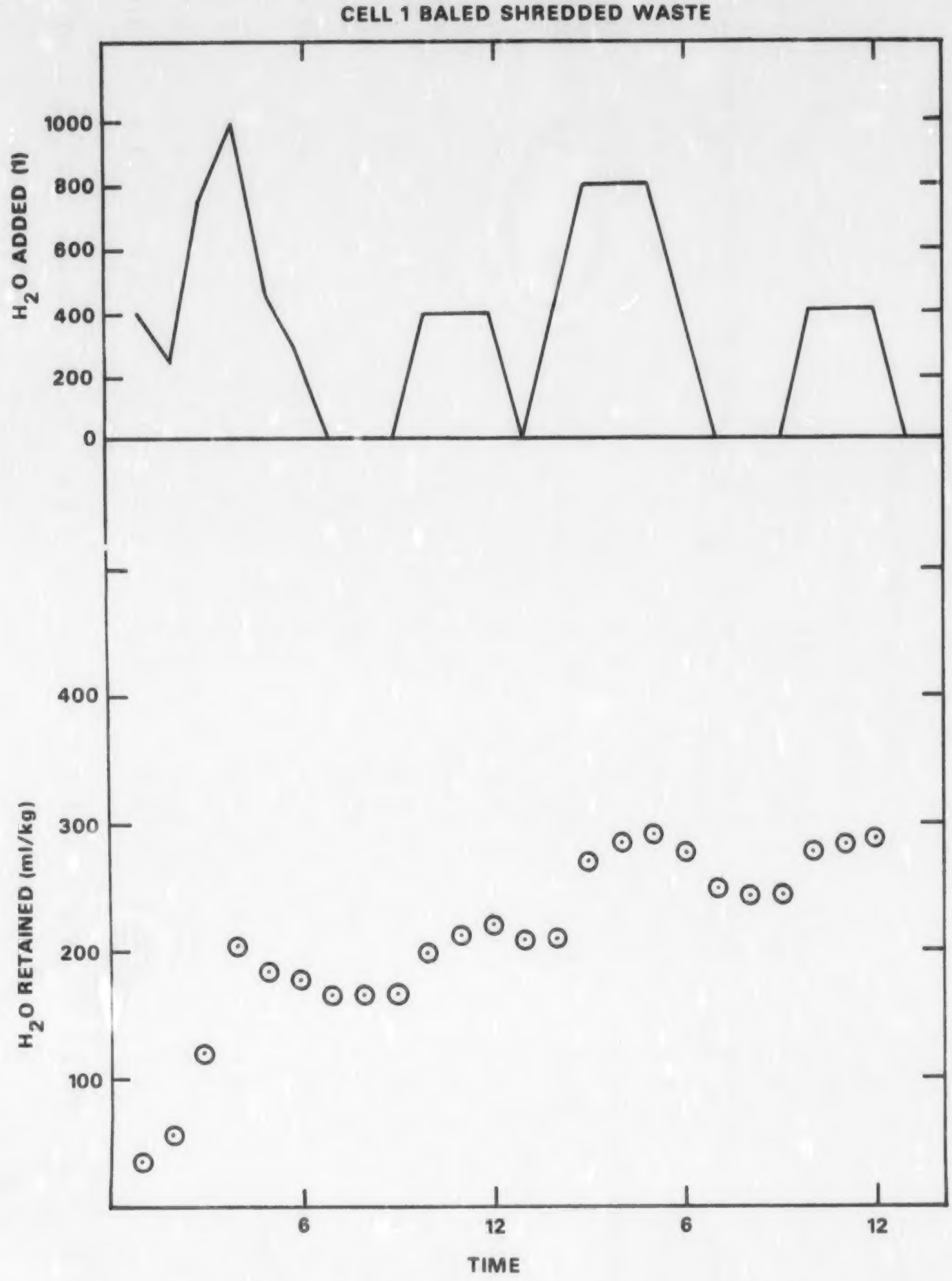

FIGURE 2. MOISTURE DATA FOR CELL1

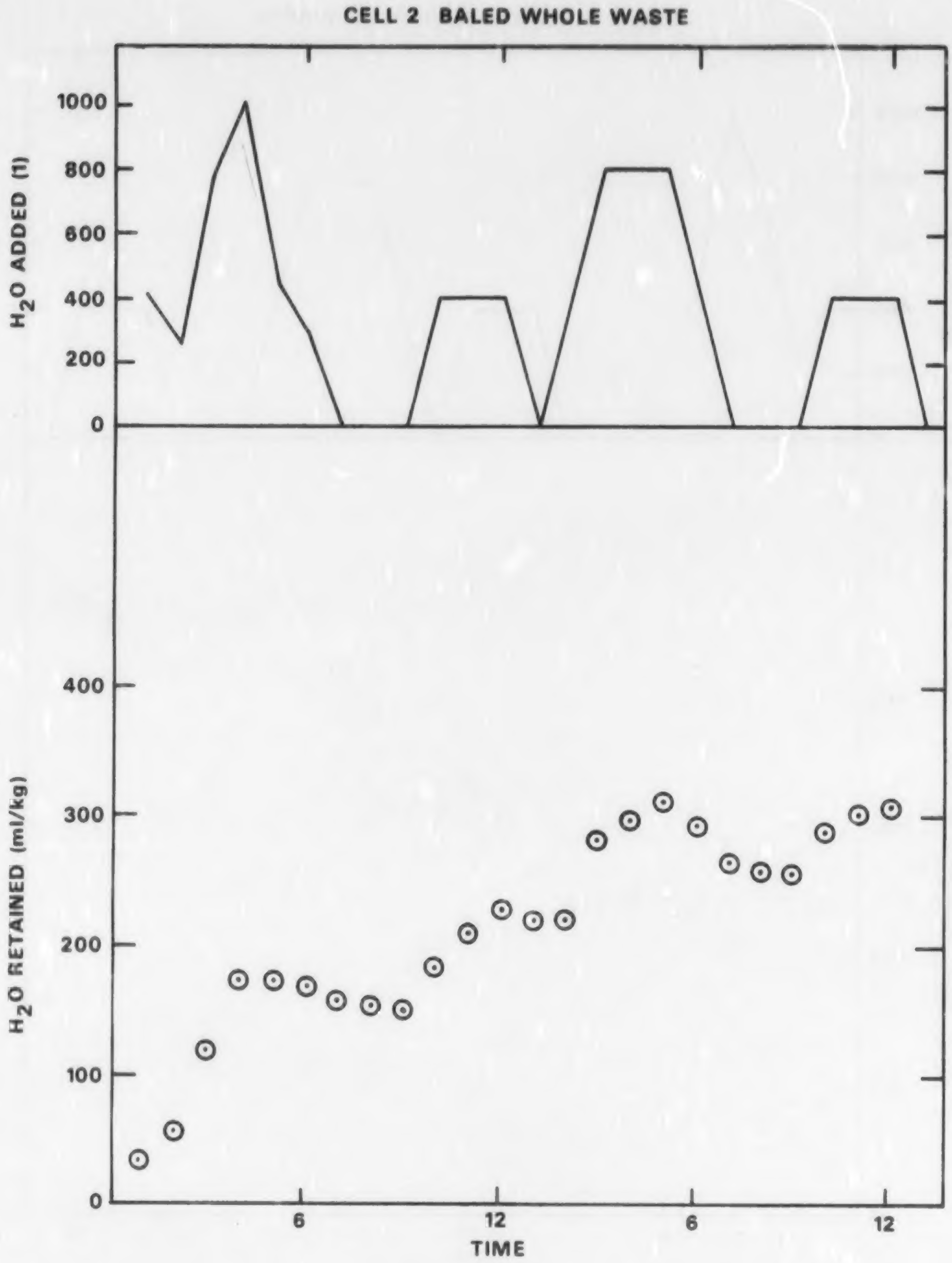

FIGURE 3. MOISTURE DATA FOR CELL 2

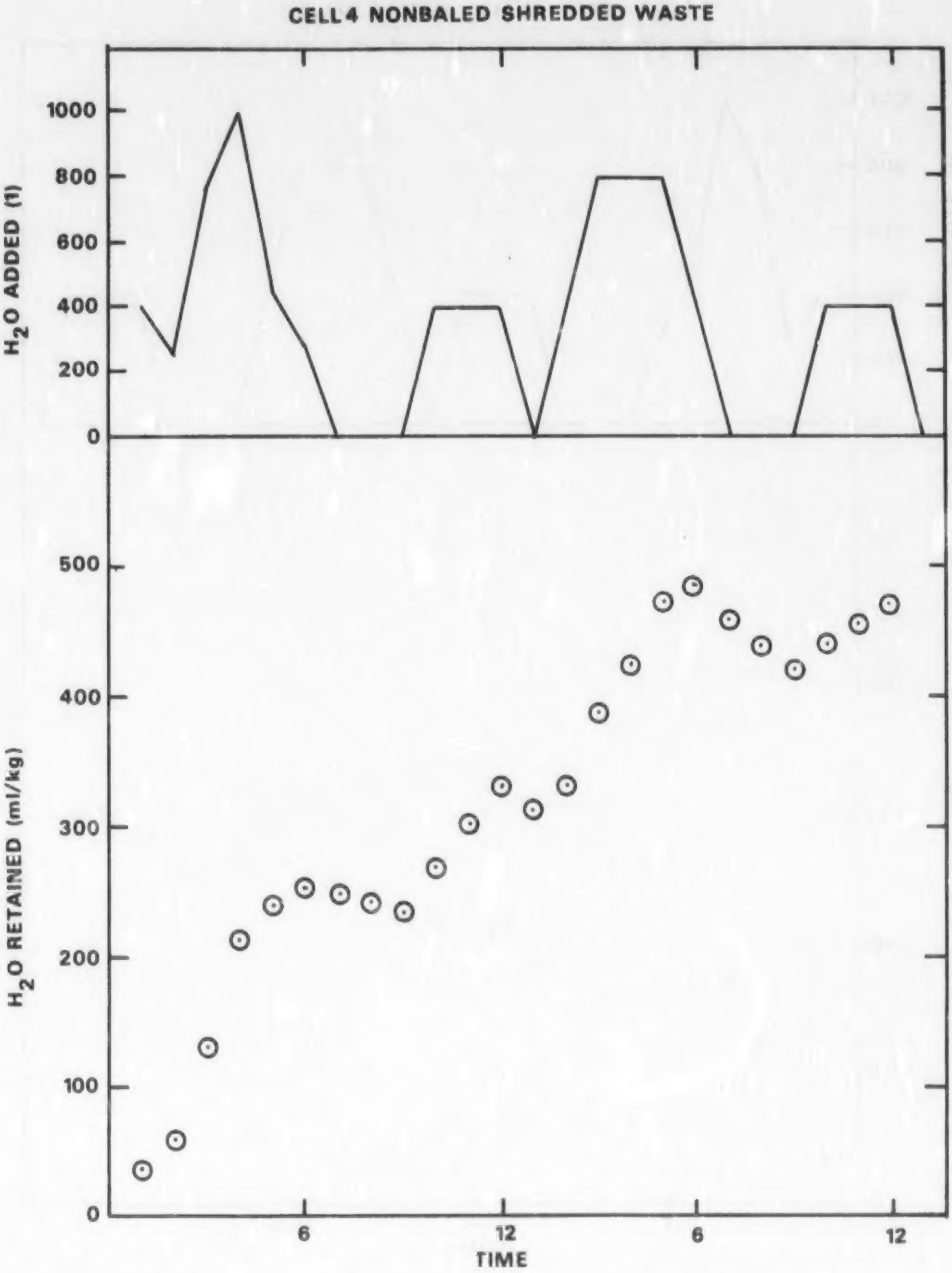

FIGURE 4. MOISTURE DATA FOR CELL4

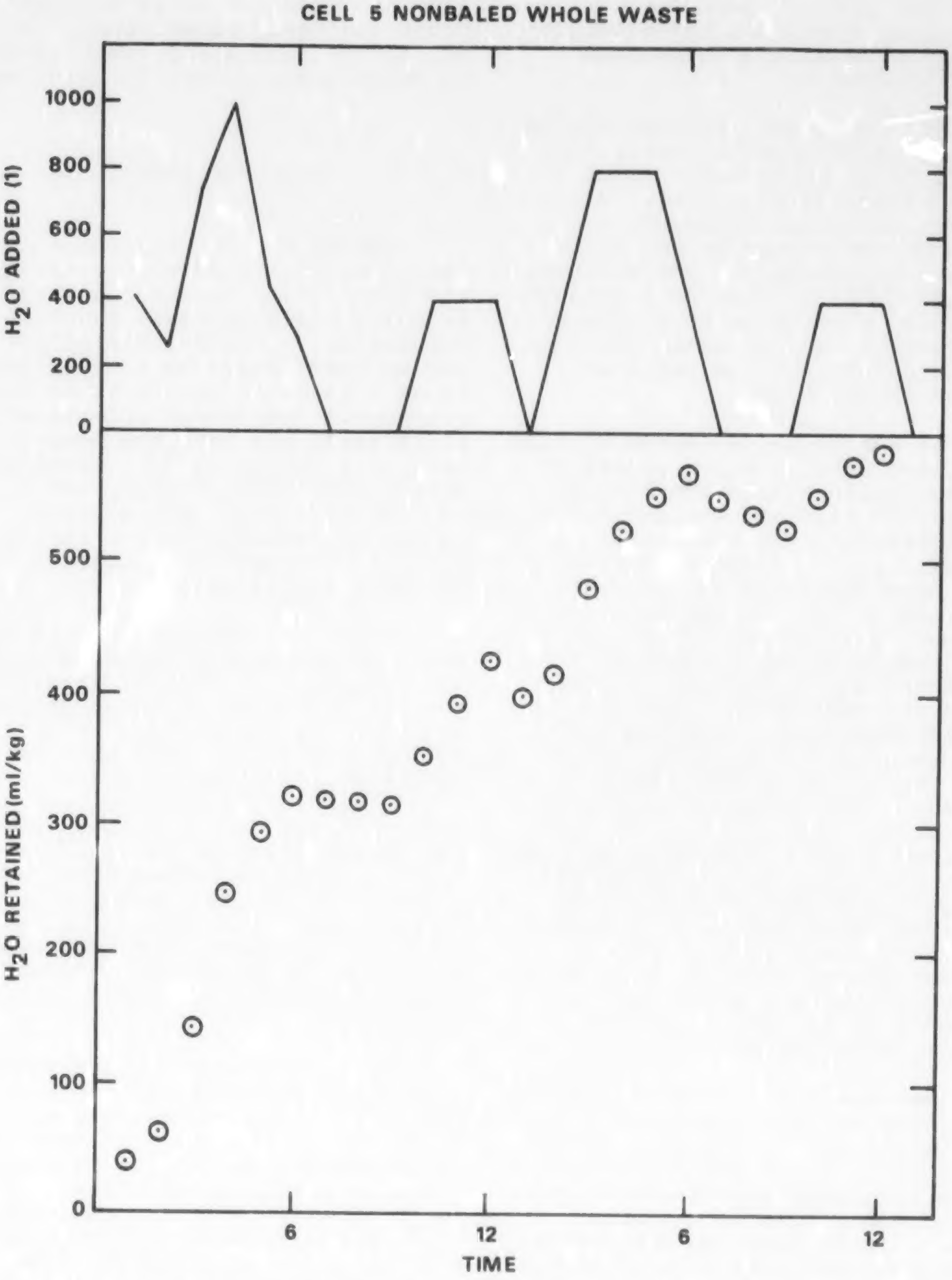

FIGURE 5. MOISTURE DATA FOR CELL 5

water. This would affect the apparent electrical resistance of the gypsum block probes. It is sufficient for our purposes now, however, to know that the moisture levels are consistent throughout each test cell.

Total moisture retained within the cell is inferred directly from the volume of water added minus the volume of leachate removed. Since the test cells are enclosed, transpiration and runoff do not enter into the calculation. The moisture retained plots in Figures 2 through 5 indicate that the rate of change of retention is decreasing but that field capacity has not yet been reached in any case.

Gas was to be collected by nine perforated plastic pipes placed in the test cells within the body of the solid waste, and one perforated pipe located in the head space of each test cell. These perforated pipes were connected to manifolds which passed through the walls of the test cell and led to the instrumentation cell. They were valved so that individual levels within the test cells could be sampled separately. Samples were to be drawn off into glass sample tubes for gas chromatographic analysis. The analysis was intended to determine the O_2, CO_2, CH_4, and H_2 contents of the product gas. Gas volume was to be determined by combining all the gas loads from an individual test cell into one line, which vents through a precision wet test meter. Further details on the gas collection system and its related problems are discussed in the following section.

Leachate was collected from the bottom of each test cell by lines which led to the instrumentation cell. The pH and conductivity of the leachates were measured and the individual leachate samples collected were measured for chemical oxygen demand, total organic carbon, alkalinity, hardness, total phosphorous, Kjeldahl nitrogen, total solids, dissolved solids, chlorides, and for the following metals: iron, copper, zinc, cadmium, chrome, lead, and nickel. Leachate samples were also analyzed for total coliform count and fecal strep count. Results of the leachate study are presented in following sections of this report.

GAS MONITORING

One of the principal goals of the study was to determine volume and composition of gas generated by solid waste in these landfill simulators. Considerable effort was expended throughout the study in an attempt to obtain valid and meaningful gas data. Unfortunately, no reliable gas data has been generated by the study. Problems encountered were several, and they will be presented here so that others may benefit by our experiences in attempting to measure gas produced in lysimeters.

First, it should be noted the wet test gas meters employed were not adequate for directly measuring the small volumes of gas produced in the early stages of a landfill lysimeter's life. When this was realized, a collection system employing impermeable gas bags for each test cell was devised. The gas produced was to be collected and once daily would be drawn off through the wet test meter by a vacuum pump. This technique is basically sound and has been used successfully by SYSTECH in laboratory studies of gas produced by anaerobic digesters.

By referring to Figure 1, the reader can see there are several points at which the structural integrity of the landfill test cell is breached. The 10 gas collection lines pass through the front wall individually; the water addition line passes through the front wall; the instrumentation access line passes through the front wall; and, finally, the leachate collection line passes through the front wall.

Each of these exit points was originally sleeved and gasketed with RTV and was thought to be gas-tight at the time.

The cell cover is a precast concrete slab. Both it and the cell wall tops are sufficiently nonuniform that the vinyl tubing used for gasketing was insufficient. Finally, the manhole in each of the cell covers was insufficiently gasketed. Each of these points is a potential gas leak and in fact and in practice, nearly every one of these points proved to be a source of gas leaks. Attempts were made to seal the cell from the outside when these leaks were discovered and the inadequacies of the design were understood.

It was discovered at that time that several purported sealant and adhesive materials were neither sealants nor adhesive. An extensive study was then undertaken by SYSTECH to determine the relative permeability of various candidate materials for use in a gas collection system. In this study, it was discovered that most plastic and rubber materials were permeable to methane and carbon dioxide, and these included several materials which manufacturers assured us were impermeable. The only adhesive sealant material which we have found to be impermeable to these gases is a 3M metal sealant.

It is our recommendation that any gas collection system used in landfill lysimeters be constructed with metal gas distribution lines, and that the gases be collected in aluminized plastic or Tedlar gas bags. Separable surfaces should be smooth and preferably should have an incised notch to ensure positive placement of any gasket material. Surfaces to be gasketed should be drawn together, preferably with a nut and bolt type arrangement. And finally, it is questionable whether concrete is a suitable material for landfill lysimeters when gas data is of interest. As a large lysimeter settles in the earth, the concrete could crack and thus destroy any gas-tight integrity which it may have had at the beginning. Large steel containers could be continuously welded for greater assurance of gas-tight integrity.

In spite of extensive efforts to seal and leak test the lysimeters in the ground at Franklin, we have come to the conclusion now that this is impossible. Effective sealing was accomplished for short periods throughout the study and some gas volume data was generated during those periods. However, the possibility of intermittent failures of gas-tight integrity must lead us to conclude that none of our gas data is reliable. We must include in this disclaimer the measured composition data since all test cells show some oxygen content. The presence of oxygen in the test cells two years after their initiation would be highly unlikely if the test cells were truly sealed.

LEACHATE DATA

The most significant results of the present study arise from the conclusions we are able to draw from the leachate data. Leachate samples were withdrawn from each test cell monthly. The volume was recorded and the various parameters mentioned previously were measured. In order to compare the effect of the various processing modes studied on the leachate produced, it was decided to do a cell-to-cell comparison of the time histories of the parameters measured. A correlation analysis comparing the results of each cell with every other cell was performed using commercially available computer programs. Correlation analysis is a statistical technique for determining the degree of association between variables. It entails calculating a correlation coefficient which is defined as covariance of the two variables divided by square root of the product of the variances of the individual variables. If the two variables are highly related, the correlation coefficient will be close to 1.0. If they are not well related, the correlation coefficient will be close to zero. Depending upon the number of times the two variables of interest are measured, the value of the correlation coefficent needed to be confident to a predetermined degree (95 percent, 99 percent, etc.) that the variables are related can be determined. It was decided in the present study to use a 95 percent confidence level as a rejection criterion. Correlation coefficients were then calculated for the leachate data using a packaged computer program, "BMDO2."

The results of this analysis were that in nearly every case, time histories of the pollutant concentrations showed better than 95 percent confidence that the data correlated from cell-to-cell. Table 2 shows the exceptions to this, i.e., those cell pairs for particular parameters which do not show 95 percent probable correlation. What this means is that while the magnitudes of the pollutant emitted from individual cells may differ, the time histories of these emissions were very similar.

In order to determine the total pollutant emitted from the individual cells, cumulative values of the mass flows of the individual parameters were calculated and plotted (Figures 6-9). The shape of the curves in each case is very similar, and would appear to be described by a power curve where the total pollutant is a function of time to the power 3 or greater. As a final check, to compare the individual cells, the total mass flow of each pollutant leached per unit mass of solid waste in the individual cells was calculated for total solids, dissolved solids, chemical oxygen demand, total organic carbon, hardness, iron, copper, zince, cadmium, chromium, lead and nickel (Table 3). Results tend to show quite similar results for each cell with Cell 5 the non-baled, non-processed waste, generally showing the lowest total pollutant mass leached to date. Even so, it does not appear to be significantly different from the leachate of the other cells.

The conclusion we draw at this time is that no significant difference can be seen in the leachate emanating from the test lysimeters no matter what processing technique is used. Two years of study is too short to determine that there is no difference in the leachate from baled or shredded or whole waste landfills, but there would appear to be no significant difference in the early stages of landfill life.

TABLE 2. COMPARISON OF TIME HISTORIES OF POLLUTANT CONCENTRATONS FROM EACH CELL

All parameters show better than 95% correlation when compared from cell to cell except as noted below.

Parameter	Cell Pairs Which Don't Correlate
pH	1-2, 1-3, 2-4, 2-5, 3-4, 3-5
Conductivity	1-3
COD	1-4
TOC	3-5
Alkalinity	
Hardness	
Phosphorous	
Kjeldahl Nitrogen	
Total Solids	
Dissolved Solids	
Chloride	1-3, 1-5, 2-3, 2-5
Iron	
Copper	
Zinc	1-2, 1-3, 1-4, 1-5
Cadmium	1-3, 1-4, 2-3, 2-4, 3-5
Chronium	
Lead	1-4, 1-5, 1-4, 2-5, 3-5, 4-5
Nickel	
Total Coliform	Only 4-5 correlate, no others do
Fecal Strep	Only 1-4, 2-3, 4-5 correlate, no others do

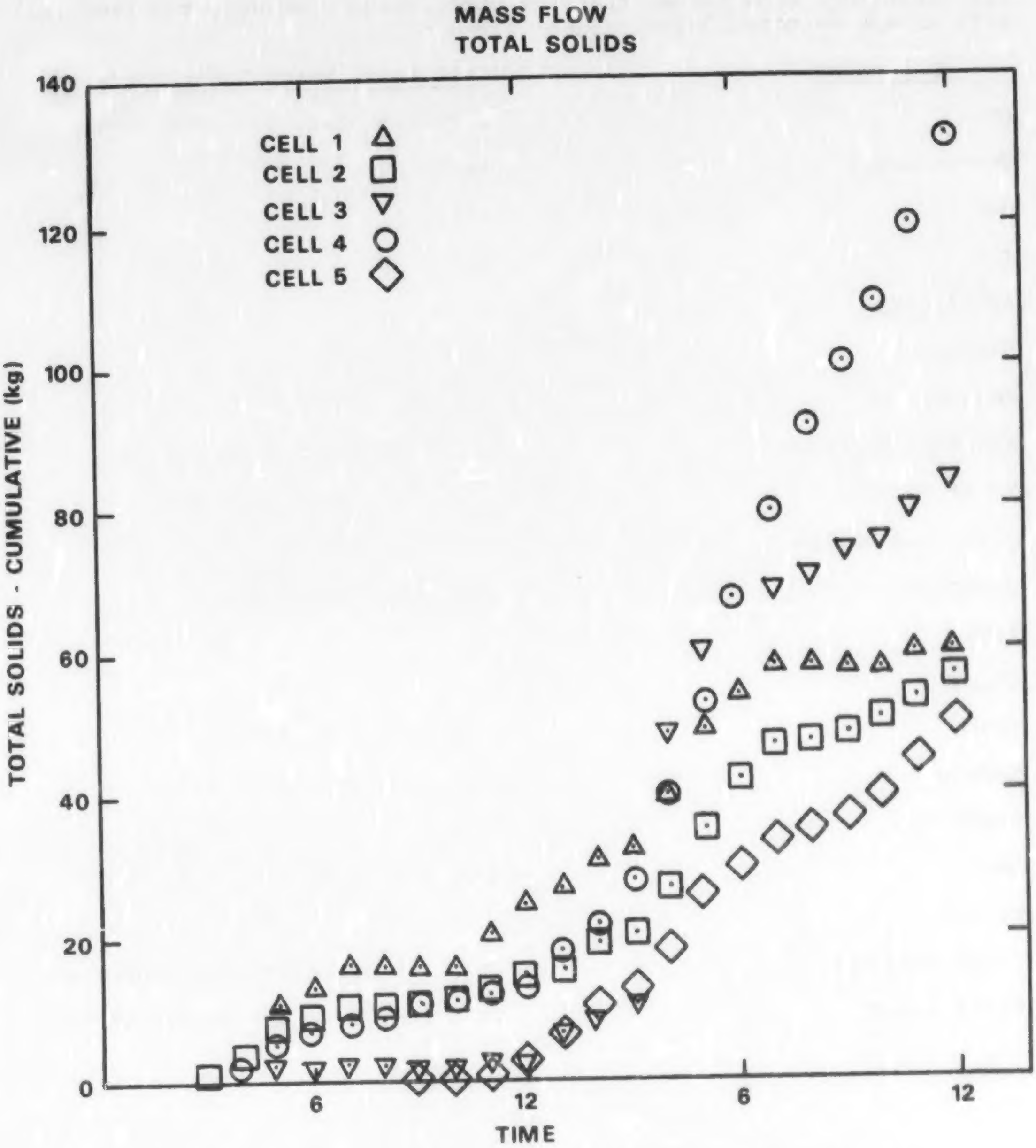

FIGURE 6. MASS FLOW TOTAL SOLIDS

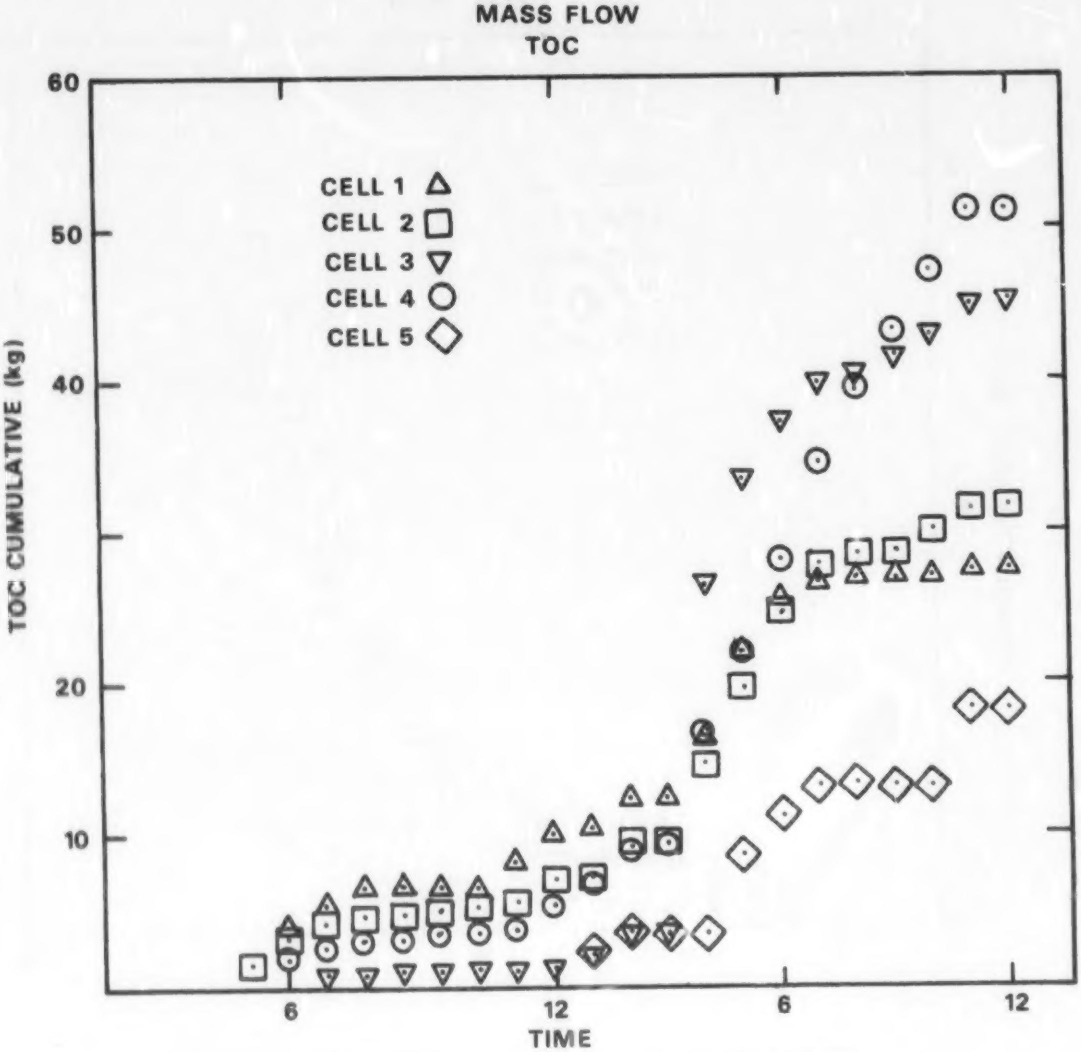

FIGURE 7. MASS FLOW TOC

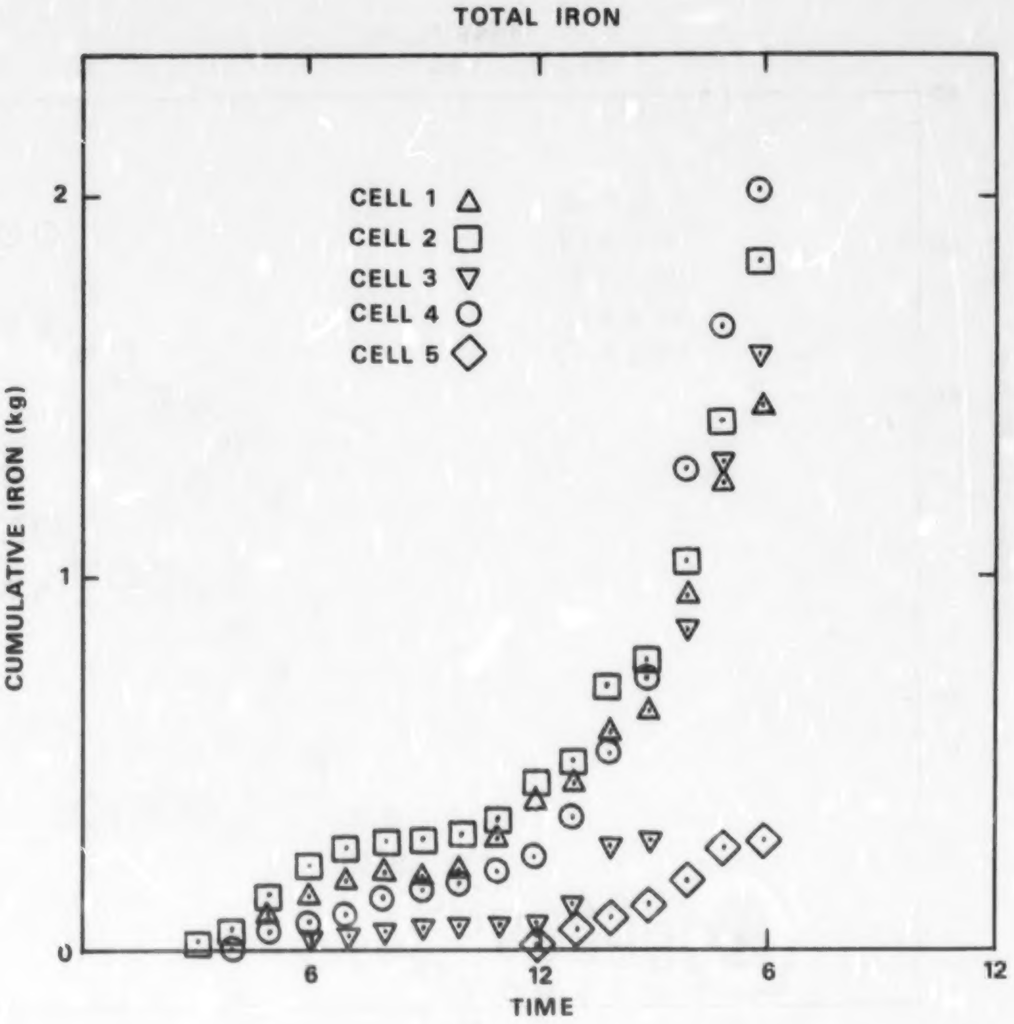

FIGURE 8. MASS FLOW - IRON

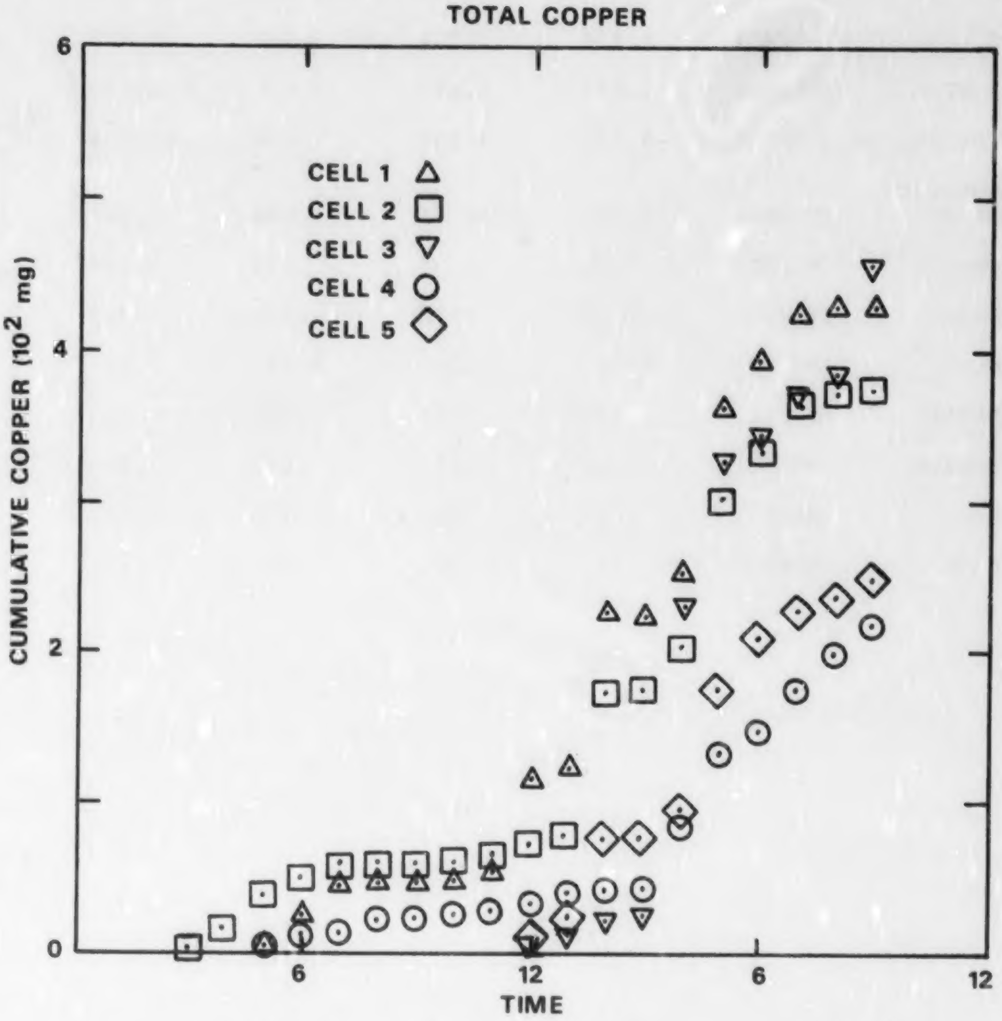

FIGURE 9. MASS FLOW - COPPER

TABLE 3. TOTAL POLLUTANT MASS LEACHED TO DECEMBER 1976

(Mass of Pollutant Per Unit Mass of Solid Waste)

PARAMETER		CELL 1	CELL 2	CELL 3	CELL 4	CELL 5
COD	(MG/KG)	5,783	7,161	10,900	15,570	7,527
TOC	(MG/KG)	2,438	2,731	3,869	4,748	1,900
HARDNESS	(MG/KG)	1,880	1,593	2,023	2,919	1,245
TOTAL SOLIDS	(MG/KG)	5,229	4,954	7,218	12,230	5,142
DISSOLVED SOLIDS	(MG/KG)	1,501	1,016	3,048	1,729	631
IRON	(MG/KG)	125	157	134	188	31
COPPER	(MG/KG)	.037	.032	.039	.020	.026
ZINC	(MG/KG)	4.58	6.59	3.18	2.12	4.12
CADMIUM	(MG/KG)	.017	.016	.020	.022	.015
CHROMIUM	(MG/KG)	.092	.103	.110	.093	.043
LEAD	(MG/KG)	.124	.095	.057	.147	.056
NICKEL	(MG/KG)	.24	.29	.37	.62	.24

EFFECT OF MOISTURE REGIME AND OTHER FACTORS ON MUNICIPAL SOLID WASTE STABILIZATION

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ABSTRACT

The present study measured the rate of solid waste stabilization under anaerobic conditions as reflected by the rate of gas production. The major gases produced by the solid waste were CO_2, H_2 and CH_4. The effect of different environmental variables on the rate of solid waste stabilization was measured. Higher rates of stabilization resulted from higher moisture contents, increasing temperatures, decreasing solid waste size and decreasing density. The H_2 was primarily detected in the containers with the lowest moisture content, while CH_4 was found in the container with the unshredded refuse.

INTRODUCTION

Placement of solid waste in landfills is the most common method of ultimate disposal of the refuse. Some impairment of the environment, however, can occur by the release of explosive gases resulting from decomposing refuse. On the other hand, these gases, primarily methane (CH_4) and hydrogen (H_2) can be recovered from the landfill and converted to useful energy by combustion. If all the methane from solid waste maintained under optimum digestion conditions could be recovered, up to 11% of U.S. natural gas demand could be satisfied (1). The formation of gaseous products parallels the decomposition of the solid waste. Since ultimate use of a landfill requires a sufficient degree of biological and physical stabilization, knowledge of the relation between the gas formation and solid wastes stabilization may allow the former to be used as a tool for landfill monitoring. It was therefore the purpose of the present study to measure the amount and composition of gases generated during anaerobic decomposition of solid waste in a simulated landfill, and to determine the effect of environmental conditions, such as temperature moisture content, size of the solid wastes and density on the quantity and quality of the product gas.

ANAEROBIC DEGRADATION OF SOLID WASTE

Formation of gases from solid waste only occurs under anaerobic conditions; the anaerobic degradation of carbohydrates, for example, does not have a net production of gas:

$$C_6H_{12}O_6+6O_2 \rightarrow 6CO_2+6H_2O \qquad (1)$$

When fatty acids such as stearic acid are degraded aerobically there is even a net reduction (31% in the case of stearic acid) in the gas phase:

$$C_{18}H_{36}O_2+26O_2 \rightarrow 18CO_2+18H_2O \qquad (2)$$

Under anaerobic conditions, however, the gas production is substantial in the acid fermentation stage in which the carbohydrates are converted to acetic and butyric acid and to a lesser extent into propionic acid:

$$C_6H_{12}O_6 \rightarrow C_2H_4O_2+C_3H_6O_2+CO_2\uparrow+H_2\uparrow \qquad (3)$$

$$C_6H_{12}O_6 \rightarrow C_4H_8O_2+2CO_2\uparrow+2H_2\uparrow \qquad (4)$$

whereas in the methane fermentation stage in which the free volatile fatty acids are converted into methane and carbon dioxide:

$$C_6H_{12}O_6 \rightarrow 3CO_2\uparrow + 3CH_4\uparrow \qquad (5)$$

The overall reaction can be described as:

$$C_rH_wO_xN_yS_z + (V - \frac{W}{4} - \frac{X}{2} + \frac{3y}{4} + \frac{Z}{2})H_2O \rightarrow$$
$$(\frac{V}{2} + \frac{W}{8} - \frac{X}{4} - \frac{3y}{8} - \frac{Z}{4})CH_4 + (\frac{V}{2} - \frac{W}{8} + \frac{X}{4} + \frac{3y}{8} + \frac{Z}{4})CO_2$$
$$+ (y)NH_3 + (Z)H_2S \qquad (6)$$

Most sanitary landfills are known to produce CO_2 and CH_4, indicating that anaerobic conditions are maintained in the solid wastes.

The hydrolysis of cellulosic materials, the major component of the organic fraction in solid waste, occurs through an enzymatic step in which a C_1 enzyme converts the crystalline cellulose to an amorphous hydrated polyanhydroglucose, which is further hydrolized by C_x enzymes to cellobiose which can be taken up by bacterial cells and converted to glucose by β-glucosidase. Cowling and Brown (2) indicated that environmental factors, especially moisture content, determine the rate of the overall hydrolysis reaction, since the moisture helps to swell the capillaries of the fibers beyond their diameter, as small as 25 Å for pulp, to allow penetration of cellulose enzymes with equivalent sizes of 30 x 170 Å. Stone *et al.* (3) therefore noted that the cellulose hydrolysis rate was proportional to the surface area accessible to a molecule of 40 Å in diameter. Thus increasing the surface area by milling of the solid waste will enhance the cellulose hydrolysis. Using the digestion efficiency of sewage sludge as a reference, Klein (4) noted that the efficiency of garbage digestion, expressed as liters of gas produced per kg solids destroyed, was 90% of that of sewage sludge. The digestion efficiency of Kraft paper was 95%, newspaper 44%, garden debris 73%, and wood only 4% of that of sewage sludge. The higher digestion efficiency of newspaper pulp as compared to wood is due to the mechanical pulping process which increases the surface area from 1 m^2/g to 40 m^2/g in pulp. The presence of lignin is thought to decrease the accessible cellulose surface area. The digestibility of pulp therefore increases from 44% to 95% after chemical removal of the majority of the lignin by sulfite pulping.

Other environmental factors such as pH, nutrient conditions and metal content, exert significant effects on the rate of cellulose digestion. Dubos (5), for example, noted an optimum pH of 6.5 to 8.2 for cellulose decomposing microorganisms. Heukelekian (6) studied the decomposition of cellulose in limed and unlimed anaerobic digesters. In the latter unit the pH decreased to 5.2 after 7 days corresponding to a 30% cellulose removal. In the limed unit the pH stabilized at 7.4 resulting in a 73% cellulose removal, indicating that higher pH values enhance the cellulose hydrolysis. Hazeltine (7) investigated the digestion of garbage in anaerobic digesters and indicated that digestion of garbage alone produces acidic conditions, but found that ratios of 1.5 to 3 parts garbage to 1 part of sludge resulted in sufficiently high pH values. Golueke and McGauhey (8) noted that the pH decreased below 6.8 when the ratio of garbage (vegetable trimmings) to sewage sludge increased beyond 3:1 in digesters maintained at a 30-day detention time. Gradual acclimation to garbage, however, allowed a successful digestion at a 100% garbage level without significant pH decreases.

Maki (9) studied the hydrolysis of cellulose in an aerobic digester seeded with anaerobic digester supernatant and noted that the primary products were H_2 and CO_2 gas and acetic acid (41% on a molar base), ethanol (39%), formic acid (15%) and lactic acid (6%). Alcohols can further be oxidized to fatty acids with CO_2 acting as the hydrogen acceptor to form methane. Mixed anaerobic bacterial strains generally resulted in higher hydrolysis rates of up to 0.07% per day. Pure cultures often experiencing hydrolysis rates of only 0.03% per day. The higher rates of the mixed cultures were due to the more rapid removal of inhibiting metabolites. Hill (10) investigated the digestion of paper pulp mixed with different amounts of sewage sludge so as to vary the C:N ratio, and noted that optimum digestion occurred at C:N ratios of 20:1 to 53:1. Heavy metals can cause inhibition of the cellulose decomposition, and Uesaka (11) measured inhibiting concentrations of 500 mg/l Fe, 50 mg/l Zn, and 5 mg/l Cu.

Several investigators noted that the cellulose hydrolysis rate is often the overall reaction rate-controlling factor, and conditions should be optimized with respect to this step. Based on the rates of removal of the metabolites, Maki (9), for example, concluded that this step was not rate-limiting and that the preceding cellulose hydrolysis rate actually determined the overall reaction rate. Chan and Pearson (12) investigated the anaerobic digestion of kraft paper pulp powder at detention times of 10, 15, 20 and 30 days and noted that about half of the carbon input was converted to gas. They further noted, similarly to Maki (9), that the hydrolysis of cellulose to cellobiose was the overall rate-limiting step.

Pfeffer (13) calculated a maximum theoretical gas production of 0.55 ℓ/g volatile solids (VS) added but actually measured a rate of 0.2 g ℓ/g VS added, or 0.24 ℓ/g solids added, at 35°C, and a 30-day detention time, indicating that only half of the volatile solids in refuse are digestible. Cooney and Wise (14) measured a gas production of as high as 0.47 ℓ/g VS added or 0.27 ℓ/g solids in digesters maintained at a 30-day detention time at 39°C.

ANAEROBIC PROCESSES IN LANDFILLS

In actual landfills similar anaerobic decomposition processes will occur. When the oxygen initially present in the gas phase within the solid waste decreases to low levels, anaerobic processes gradually replace aerobic processes. This is also reflected by the increase of the CO_2 content above 20%. Only 20% can ever be found under aerobic conditions, as only that much CO_2 can be found from the consumed O_2 present at an initial concentration of 20%. Anaerobic conditions generally restrict the activity of the aerobic thermophilic organisms and enhance the mesophilic anaerobes. This shift is generally accompanied by a decrease of the landfill temperature. Eliassen (15), for example, noted that aerobic conditions prevail near the surface of landfills resulting in relatively high temperatures (71°C at 0.9 m depth), while in deeper layers anaerobic conditions and lower temperatures prevail (40°C at 3.3 m). Anaerobic processes are generally occurring at a lower rate than aerobic processes and landfill stabilization is therefore more rapid under aerobic conditions. The substantially lower moisture contents, in landfills as compared to anaerobic digestion studies discussed earlier will result in far lower gas production rates. Carpenter and Setter (16), for example, noted that in a dry fill the gas phase consisted of 34% CO_2 and 7% CH_4, while in a moist landfill the concentrations were as high as 35% and 65%, respectively. Merz and Stone (17) similarly noted an increase in the methane content with increasing moisture of the solid waste at the Spadra test sites. The existence of anaerobic processes was further confirmed by Bishop et al. (18) who measured an excess gas pressure in landfill which forced the gases out through the surface cover (93-95%) or through the bottom and side walls of the landfill (5-7%). They further calculated a maximum gas production rate of 0.128 ℓ/kg.day during the initial CO_2 bloom. An increase in moisture content as a result of rainfall entering the fill caused a second CO_2 bloom and a rise in the gas production to 0.099 ℓ/kg. day. During the three-year observation period the methane content increased to about 11%. Rovers and Farquhar (19) noted that excessive infiltration of melt water decreased the methane content in the gas phase from 19% to 4%, while the CO_2 content increased and pH of the leachate decreased, indicating that the acid hydrolysis under such conditions is occurring at a faster rate than the methane fermentation. When stable conditions exist in the landfill the opposite is observed. Merz and Stone (20), for example, noted that the instantaneous application of 11 to 24 cm of water caused a significant increase in the methane content of the gas phase in favor of the test cells at the Spadra landfill.

Very few studies measured the quantity and composition of gases produced when solid waste was placed under anaerobic conditions in an enclosure. This approach is by far the most accurate way to measure the gas quantity, and was successfully used by Merz (21), Merz and Stone (17), Ramaswamy (22), Rovers and Farquhar (19). Jackson (23) and McCabe (24) are using a similar technique with industrial waste and baled solid waste; the present study was funded by the U.S. Environmental Protection Agency parallel to the latter two studies and intended to investigate some of the same factors that influence the gas production generated during anaerobic decomposition of solid waste.

MATERIALS AND METHODS

The present study employed eighteen 208 ℓ (55 gallon) steel containers that were filled with 55 to 80.5 kg dry weight of solid waste, sealed and maintained at different environmental conditions, with subsequent measurement of the gas production and composition. A representation of the solid waste container is shown in Figure 1. Each drum was lined with 10 polyethylene bags of 0.15 mm thickness. A cushion of construction sand was placed between the drum floor and the plastic liner, with a gradual slope to a height of about 8 cm at the edge of the drum to funnel leachate towards the drain. A 15-cm layer of Class A gravel provides collection of the leachate and screening of the drain. A fitting was installed on the top of the drums to allow the application of water to the solid waste (Figure 2, Detail B), while a second fitting allowed for collection of gases and access for the thermocouple wires (Figure 2, Detail A). The drain installed at the bottom of the container allowed for collection of leachate (Figure 3, Detail D).

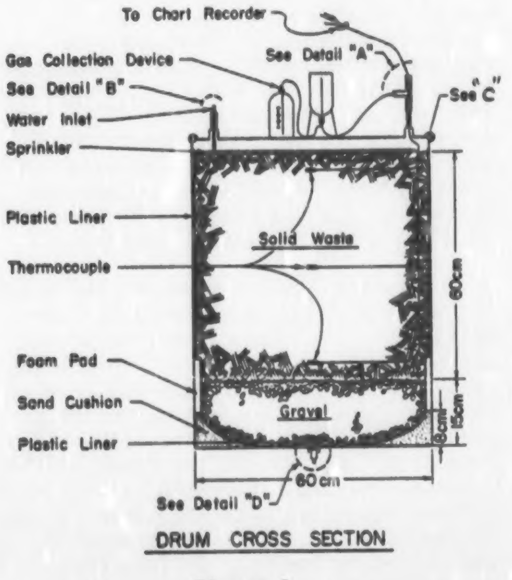
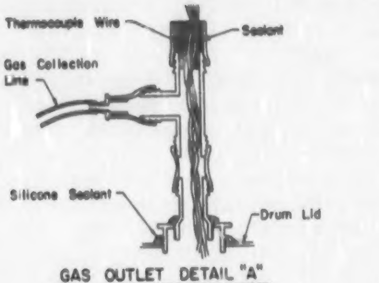

To Chart Recorder
Gas Collection Device
See Detail "A"
See Detail "B"
See "C"
Water Inlet
Sprinkler
Plastic Liner
Solid Waste
Thermocouple
60cm
Foam Pad
Sand Cushion
Gravel
15cm
8cm
Plastic Liner
60 cm
See Detail "D"
DRUM CROSS SECTION

Figure 1
Cross-section of Solid Waste Container


Thermocouple Wire
Sealant
Gas Collection Line
Silicone Sealant
Drum Lid
GAS OUTLET DETAIL "A"

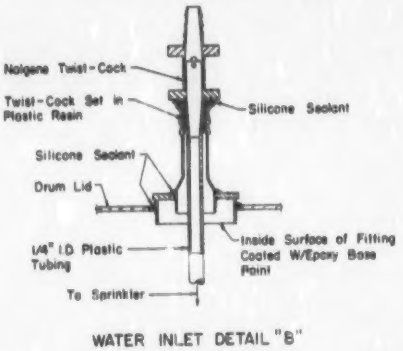
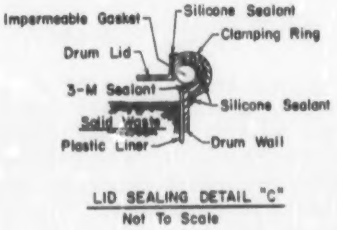

Nalgene Twist-Cock
Twist-Cock Set in Plastic Resin
Silicone Sealant
Silicone Sealant
Drum Lid
Inside Surface of Fitting Coated W/Epoxy Base Paint
1/4" I.D. Plastic Tubing
To Sprinkler
WATER INLET DETAIL "B"

Figure 2
Details of Gas Outlet and Water Inlet


Impermeable Gasket
Silicone Sealant
Clamping Ring
Drum Lid
3-M Sealant
Silicone Sealant
Solid Waste
Plastic Liner
Drum Wall
LID SEALING DETAIL "C"
Not To Scale

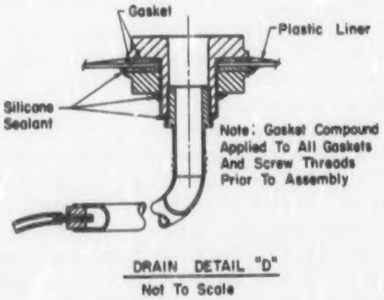

Gasket
Plastic Liner
Silicone Sealant
Note: Gasket Compound Applied To All Gaskets And Screw Threads Prior To Assembly
DRAIN DETAIL "D"
Not To Scale

Figure 3
Details of Lid Sealing and Drain

The solid waste for the study was obtained at the City Solid Waste Reduction Plant in Madison, Wisconsin. The solid waste was collected by municipal employees in wards 6, 7, 13, and 15 on Thursday January 15, 1976. Ward 6 is located in downtown Madison with values of homes generally less than $10,000. Ward 7, with old homes in the $20,000 range often occupied by students, is also located in downtown Madison. Wards 13 and 19 are both in suburban areas with homes in the $30,000 and $50,000 range, respectively. The solid waste is collected once a week and is brought to the reduction plant where it is milled by municipal employees using a Tollemache mill, and shredded to a nominal size of 0.7 to 2.5 cm. The shredded solid waste was placed in 208 ℓ drums, compacted to a density of 150 kg/m^3, transported to the University of Illinois and stored, covered, at temperatures below 0°C.

The solid waste was packed into the test containers through the use of a large hydraulic ram ordinarily used for material strength testing. A 500 kg force proved adequate in the compaction of the solid waste to a dry density of 310-436 kg/m^3. The degree of compaction was limited by the resilliency of the solid waste and considerable expansion occurred as the ram was withdrawn. After compaction, the drums were sealed. As the moisture content of the milled solid waste was as low as 28%, calculated on a dry-weight basis, water was added to provide a more representative baseline moisture content of 36%.

Sealing the drums adequately was a major problem, as several sealants such as a caulking compound, a synthetic rubber (Miracle Seal, Revere Chemical Corp., Salem, Ohio), a gasket compound (Form-a-Gasket #2, Permatex, Inc., Kansas City, Kansas) and an aluminum-backed duct tape, proved unsuccessful. The sealant combination that did work was a ketone based metal (#2084, 3M Company, St. Paul, Minnesota) and a silicone rubber sealant (#8640, Dow-Corning, Midland, Michigan) (Figure 3, Detail C). In sealing the drums, the lip of the drum lid was given a coating of the 3-M sealant and then fitted with an impermeable gasket. The gasket and drum rim were then coated with the 3-M sealant and the lid was placed on the drum and temporarily clamped in place. The part of the gasket that remained exposed then received an application of the 3-M sealant, forming a continuous layer of sealant between the drum and the lid. After this layer was cured, a process which required 2 to 3 days, it was coated with a layer of silicone rubber sealant. A heavy duty bolt-type clamping ring was then installed and tightened as much as possible. The points where the ring contacted the drum were then coated with the silicone rubber sealant. Finally, the fittings through which gas, water, and thermocouple wires pass, as well as the bottom drain, were given a coating of silicone rubber sealant. After curing for an additional two days, the drum was pressurized to test for leaks. The ability of the seal to contain a pressure of 100 cm of water for 24 hours was taken as an indication that the drum had been adequately sealed.

The gas produced in the test cells is collected using Mariotte flasks, consisting of two bottles, one of which is placed at a higher level than the other. As gas is produced, it flows into the upper bottle, displacing water from the upper bottle into the lower one. Measuring the change in water levels allows a determination of the volume of gas produced. Gases were collected intermittently rather than continuously, keeping the drums completely sealed except for brief periods during the gas collection. Gas was therefore collected from the drums when the initial pressure exceeded 2.5 cm of water, measured with a manometer attached to each drum. The amount of gas produced in actively generating containers was made with a Precision Wet Gas Meter (Precision Scientific, Chicago, IL). Gas volumes were corrected for vapor pressure of water and variation in temperature and pressure. At low gas flow rates, the wet gas meter was not activated, and Mariotte flasks were used. An extensive analysis was made of the composition of the collected gases using a Fisher gas partitioner (Fisher Comp., Pittsburgh, PA) model 25V using 30% DEHS on 60/80 mesh column pack as the first column and a Molecular Sieve 13X as the second column. Helium is used as carrier gas for the analysis of O_2, CO_2, and N_2, while argon is used for the analysis of H_2.

Temperatures in the test cells were monitored by copper constantan thermocouples placed in the test cells at the time of loading (Figure 1). All three thermocouples were centrally located in relation to the drum wall. To protect against corrosion each thermocouple was encased in heat-shrinkable tubing, which in turn was coated with a seal (Miracle Seal, Revere Chemical Corp., Salem, Ohio).

RESULTS AND DISCUSSION

Different factors were evaluated in the present study, i.e., the effect of moisture content, temperature, solid waste size, density, and exposure to air, and a summary of conditions and results is given in Table 1. The largest number of containers was used to evaluate the effect of moisture conditions on the gas production; the necessary water was added to the solid waste at once shortly before the sealing of the drum lids. The effect of temperature was evaluated by placing one container in a room with an average temperature of 26°C while all other containers were placed at 17°C. The effect of solid waste size was evaluated by filling two containers with unshredded solids waste torn by hand to approximate sizes of 12.5 cm and 25 cm, respectively. The solid waste density was evaluated in two containers with a density higher and lower, respectively, than the majority of the other containers. Three containers were exposed to air to evaluate the effect of air addition on the gas composition. The results presented herein represent the first year of data collected under steady-state environmental conditions. During the second, third, and fourth year, transient environmental condition will be evaluated by continuously adding simulated rainfall to several of the containers with the 36% moisture content. Water will be added at a rate of 25 cm/year at biweekly intervals and 50 cm/year at weekly intervals. One of the 99% containers will be evaluated for leachate recirculation while the other 99% container will receive 25 cm/year with pH adjustment.

Before the tests were started, the solid waste composition was analyzed in detail. A large sample size (245 kg) of the solid waste, collected the same day as the milled refuse was separated into the different components, dried and weighed. The relative composition of the dried solid waste is given in Table 2. Comparison with other data (25) showed that the percentage of paper was less than that generally observed. This could well be due to the fewer students present in Madison during the semester break at the University of Wisconsin. More rock, ash and dirt were also observed in the sample which could be attributed to the cold weather conditions during the week that the solid waste was generated.

After loading of the solid waste in the containers and sealing of the drums, temperature and gas measurements were started on March 17, 1976. The temperature in each container only rose slightly after the sealing of the containers, as shown in Figure 4. The difference between the room temperature and the container temperature varied between 2.2°C and 10°C. The highest temperatures were observed in the upper layers in container 17 exposed to air, indicating that aerobic decomposition is able to raise the temperatures considerably. The sealed containers, with the gas line left open, were comparable to the completely sealed ones, indicating that only a substantial exposure to air will result in thermophilic conditions. No consistent relation could be detected between the average temperature differential and moisture content, and the size of the solid waste. When the differential temperature was related to the dry density, the data tend to indicate that a higher density resulted in a lower temperature differential. The average temperature differential of the warmest 50% of the containers is 4°C corresponding to an average dry density of 357 kg/m^3. The average temperature differential of the coldest 50% of the containers is 2°C corresponding with a dry density of 396 kg/m^3.

All of the containers produced gas and the amount produced after 300 days varied between 5.6 ℓ to 465 ℓ equal to an average daily rate of 0.29 to 20.2 mℓ/kg dry weight day (Table 2). The largest amount of gas produced represents an amount of 6.0 ℓ/kg dry weight (Figure 5) which is 2.5% of the amount measured by Pfeffer (13) in anaerobic dispersed solid waste digesters. The results in Figure 5 clearly indicate that the cumulative gas production in the 99% moisture containers (numbers 6, 13, 14) increases linearly and rapidly during the first forty days after sealing while subsequent rates, often linear for extended periods, can be as low

Table 1: Results of Gas Production and Composition Measurements During Anaerobic or Aerobic Decomposition of Solid Waste

Effect Studied	Container Number	Initial Moisture Content (% d.w.)	Weight of Solid Waste (kg)	Dry Density (kg/m^3)	Temperature (°C)	Size (cm)	Average Gas Production During First 300 d. (mℓ/kg.d)	Maximum Combustible Gas Composition (%)
Moisture Condition	1	36	65.0	334	17	2.5	0.29	7%H_2
	2	36	66.4	393	17	2.5	3.50	-
	5	36	67.3	345	17	2.5	5.63	-
	7	36	66.4	368	17	2.5	0.77	12%H_2
	9	36	67.3	399	17	2.5	0.43	15%H_2
	10	60	69.1	342	17	2.5	3.87	1%H_2
	15	78	67.3	375	17	2.5	12.9	-
	6	99	67.7	386	17	2.5	17.8	-
	13	99	79.5	436	17	2.5	15.8	-
	14	99	76.8	403	17	2.5	20.2	-
Temperature	11	36	69.5	356	26	2.5	1.07	5%H_2
	(1,7, 9)*	36	66.2	367	17	2.5	0.50	7-15%H_2
Size	4	78	60.5	310	17	2.5	1.63	50%CH_4
	18	78	60.4	356	17	12.5	9.03	-
	(15)	78	67.3	375	17	2.5	12.9	-
Density	12	36	80.5	421	17	2.5	1.23	-
	(1,2, 5,7, 9)*	36	66.5	368	17	2.5	2.12	0-15%H_2
	16	36	55.0	335	17	2.5	9.7	15%H_2
Open to Air	17	36	71.4	406	17	2.5	-	-
	3**	36	70.0	375	17	2.5	-	-
	18**	28	71.4	399	17	2.5	-	-

* Average values of the containers indicated.

**Sealed but gas outlet left open.

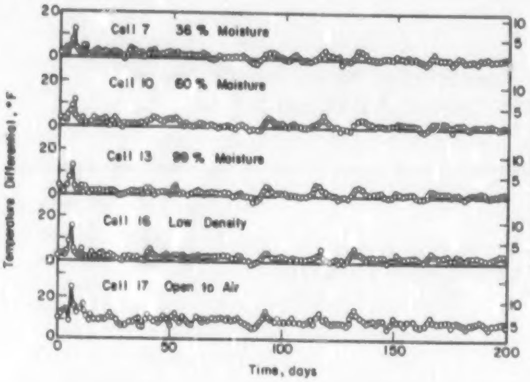


Figure 4
Temperature in Container 7, 10, 13, 16, and 17 During the First 200 Days as Averaged for the Thermocouples Located on Top, Middle and Bottom in the Solid Waste

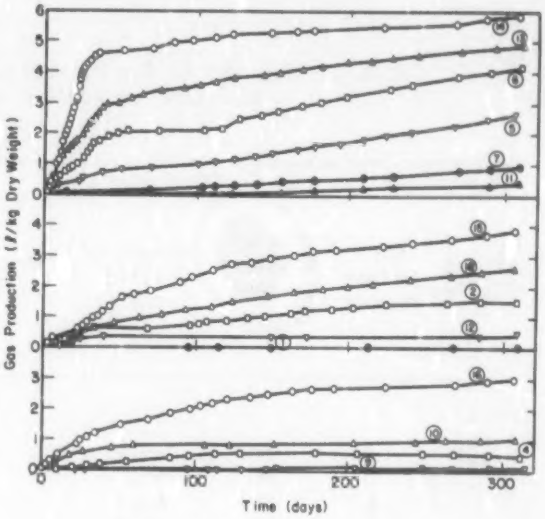


Figure 5
Cumulative Gas Production in Fifteen Containers

Table 2: Composition of the Solid Waste Before Shredding Collected on Thursday, January 15, 1976

	This Study % of dry weight	US-EPA (25) % of dry weight	Reinhardt and Ham (26) % of wet weight
food waste	14.4	11.1	15.3
garden waste	3.1	6.9	13.8
dirt	14.9	8.3	7.2
plastic	2.8	2.8	3.9
paper	36.5	48.6	46.2
metal-ferrous	11.8	11.1	6.7
metal-nonferrous	2.9	11.1	6.7
glass	6.8	8.3	10.1
textiles	0.7	0.7	1.6
wood	1.8	2.1	1.1

as 5% of these initial rates. This may indicate a zero-order reaction rate at high moisture contents. The stepwise pattern further shows that a major portion of the solid waste, varying between 30% and 75%, is rapidly degradable within a relatively short period of time. The shape of the other cumulative gas production curves generally do not show this stepwise pattern and resemble more closely a parabolic or inverted logarithmic curve possibly reflecting half- or first-order reaction rates. The lowest gas production occurred in the container with the lowest moisture content.

The collected gases of all containers were analyzed weekly or less frequently when the composition did not show large variations. The results in Figure 6 clearly show the absence of O_2 indicating that anaerobic conditions are prevalent. The N_2, initially present at 79%, is rapidly displaced by the generated CO_2 which then becomes the major component in the gas phase. The highest CO_2 content was generally observed in the containers with the highest moisture content. Comparison of the gas production rates in the high moisture content containers is due to the CO_2 "bloom." After approximately 50 days,8 of the 15 sealed containers showed hydrogen in the gas phase which amount peaked after approximately 100 days and gradually decreased thereafter. Only one container filled with large sized solid waste (Number 4) so far has experienced production of methane (Table 2) at a volume percentage of 50%. The methane appeared immediately in the container and did not show a sequential pattern in which the methane generation follows the CO_2 bloom (19). The gas composition data therefore indicate that most containers have just completed the acid hydrolysis phase characterized by CO_2 and H_2 generation and that the methane fermentation will probably start in most containers after more than a year period. Thus, although most of the containers did not produce any energy yielding methane gas in large quantities, as CO_2 was the main component in the gas phase, the results of the first phase of this study are important to assess the impact of environmental conditions on the CO_2 formation. Bishop (18) indicated that CO_2 can diffuse large distances through the soil and increase the alkalinity in groundwater which would increase the corrosion potential.

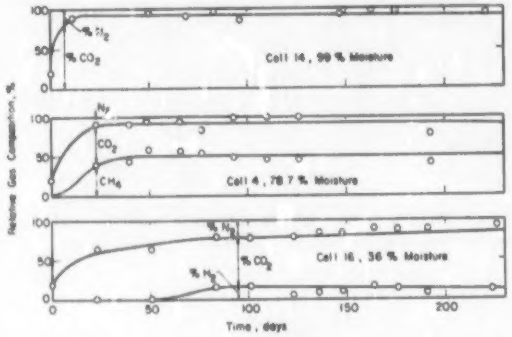


Figure 6
The Gas Composition in Three Representative Containers

A systematic evaluation, therefore, was made of the factors that influence the magnitude of the gas production during the acid digestion phase. For that purpose the average gas production rates during the first 300 days were calculated and correlated with the different environmental conditions as shown in Figure 7. One of the major factors influencing the average gas production rate is indeed the moisture content (Figure 7A). Increasing the moisture content from 36% to 99%, a 275% increase, resulted in an increase of the gas production from 2.1 to 17.9 ml/kg.d, i.e., an 852% increase. The increase is most noticeable between 60% and 78% moisture content but tended to level off at higher contents. While values averaged for several containers show a clear trend, significant variability exist among containers maintained at identical conditions. The highest gas production at the 36% moisture condition, for example, is 265% of the average value, while the lowest rate is 14% of the average gas production. At higher moisture contents the variability becomes less as the highest rate at 99% moisture is 113% of the average, and the lowest rate 88% of the average. It is further interesting to note that the

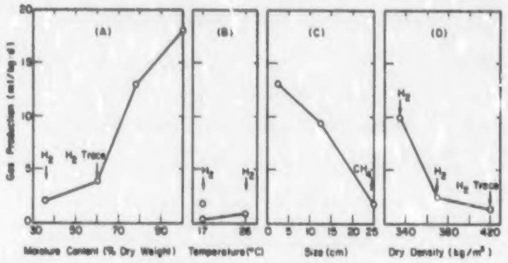


Figure 7
Relation Between Gas Production Averaged Over the First 300 Day Monitoring Period, and Moisture Content (A), Temperature (B), Solid Waste Size (C), and Dry Density (D)

hydrogen gas is only produced in the containers with a low moisture content, possibly indicating that at high moisture contents hydrogen acceptors exist. Under such condition the cellulose hydrolysis step may result in the formation of significant quantities of aldehydes or alcohols. Since a high moisture content results in a high rate of gas production, an inverse relation tends to exist between rate of gas production and maximum hydrogen content in the gas phase.

A second important factor, the temperature, also tends to increase the rate of gas production. Excluding containers 2 and 5, that did not have any H_2, it can be calculated that increasing the temperature from 17°C to 26°C (a 9°C increase) increases the rate of gas production from 0.50 to 1.07 ml/kg.d. (a 214% increase). However, if the results of containers 2 and 5 are included, obviously no temperature effect is observed (Figure 7B). Furthermore, the large variability at such low gas production rates prevents any firm conclusion.

The size of the solid waste tended to have a noticeable effect on the rate of gas production (Figure 7C). Decreasing the size of the solid waste by a factor of 10 increased the rate of gas production by a factor of 7.9, and this relation was approximately linear in the intermediate range. Of all containers evaluated, methane was only produced in the one with the large size of solid waste. The methane formation in this container may well be due to the absence of shredding which mechanical operation in effect may have mixed the solid waste so well that it dispersed bacterial nutrients to suboptimum concentration. A more likely explanation, however, is that the shredding of solid waste exposed a larger surface area of cellulosic materials thereby enhancing the cellulose hydrolysis. This results in a rapid free volatile fatty acid formation, and thus depresses the pH to values toxic to methanogenic bacteria. If this hypothesis is true, it indicates that milling of solid waste will enhance its degradation by hydrolysis and methane fermentation as long as sufficient buffer capacity is present; for example, resulting from the addition of sewage sludge.

Increasing the density of the solid waste tends to decrease the rate of gas production (Figure 7D) which is to be expected since compaction tends to decrease the effective surface area exposed to enzymatic hydrolysis. Baling of solid waste to high densities is therefore expected to result in very low gas production rates, a finding actually reported by McCabe (24).

The present study noted only methane present in the container with the large size solid waste. Using shredded solid waste fractions, Merz (21) observed a maximum CH_4 content of 0.9% even though a wide range of moisture contents was tested. Even in the large solid waste container Rovers and Farquhar (19) only noted a 2.8% methane content. Ramaswamy (22), however, noted stable methane fermentation often as little as 40 days after the initiation of the tests. Careful examination of this data (Figure 8) show that the methane content does not increase until the pH of the solid waste moisture increases beyond 5.0. The high food waste content (primarily dog food) may have resulted in a sufficient amount of NH_3 released during the degradation

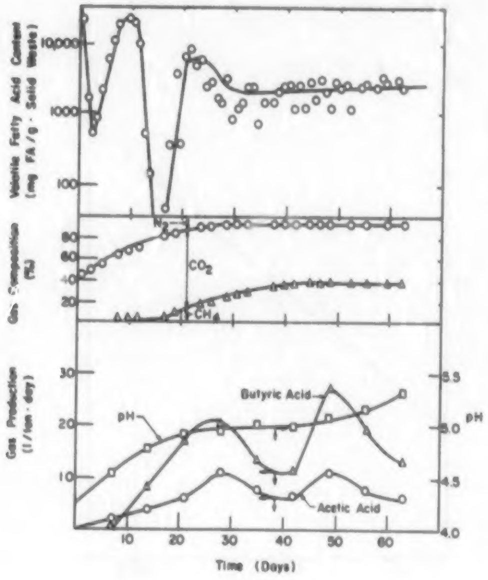


Figure 8
Rate of Gas Production, Gas Composition, pH and Fatty Acid Content as Measured by Ramaswamy (22)

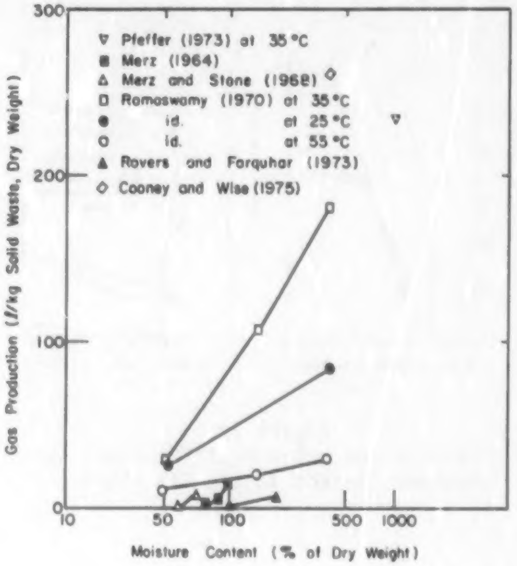


Figure 9
Effect of Moisture Content on Total Gas Production (l/kg dry weight)

of the amino acids to counteract the pH decrease resulting from the free volatile fatty acid formation. Thus, addition of strong buffering substances may well be necessary to initiate the methane fermentation in shredded solid waste, or solid waste subject to vigorous cellulose hydrolysis resulting from large water additions. Figure 8 further shows a sequential gas production pattern with an initial aerobic-anaerobic phase followed by the CO_2 "bloom" and the methane fermentation.

The importance of the moisture content on the gas production (Figure 7A) is confirmed by the result of other studies as shown in Figure 9, possibly indicating that linearly increasing gas production can be realized by a logarithmically increasing moisture content. The highest results are obviously obtained by Pfeffer (13) and Cooney and Wise (14) employing anaerobic digesters with more optimum mass transfer conditions. The moisture content not only influences the total amount of gas produced, but also determines the extent of the methane fermentation. The data by Ramaswamy (22), Merz and Stone (20) and Merz and Stone (27) indicate that increasing moisture contents increase the maximum methane content in the gas phase up to 55 to 60%. Merz and Stone (20) further noted that increasing moisture contents increased the H_2 in the gas phase. Merz and Stone (27) in later studies and Ramaswamy (22) did not note such large effect. The data of the present study show an inverse relationship between the H_2 and the moisture content (Figure 10).

The observed effect of temperature (Figure 7B) was also noted by Merz (21), as plotted in Figure 11, it shows a maximum rate near 35°C, the optimum for anaerobic mesophylic digestion. The gas production in landfills can therefore be considerably lower than maximum possible rates attainable at higher temperatures. Using the temperature data from the present study, and from Ramaswamy (22), it is possible to calculate the activation energy of the reaction as defined by:

$$\ln k_2/k_1 = E/R\ (1/T_1 - 1/T_2) \qquad (1)$$

k_1, k_2 = reaction rate contents at T_1 and T_2, respectively

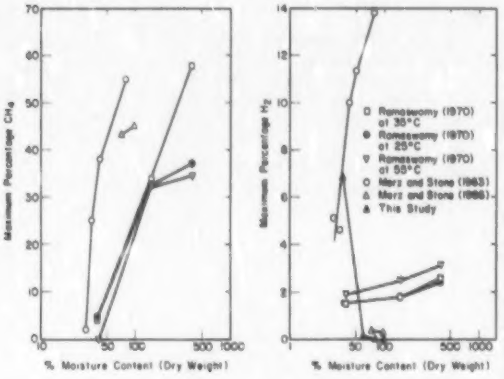


Figure 10
Effect of the Solid Waste Moisture on Methane Content in the Gas Phase

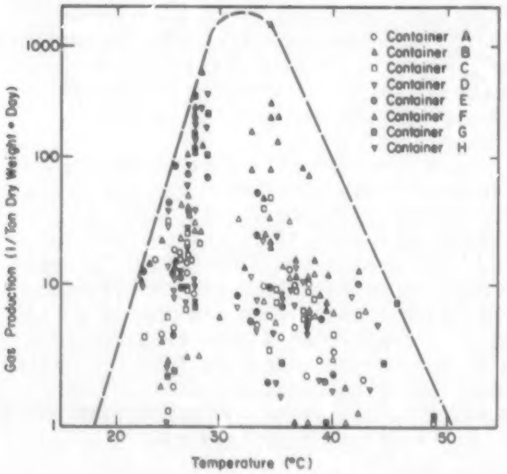


Figure 11
Effect of Temperature on the Rate of Gas Production (21)

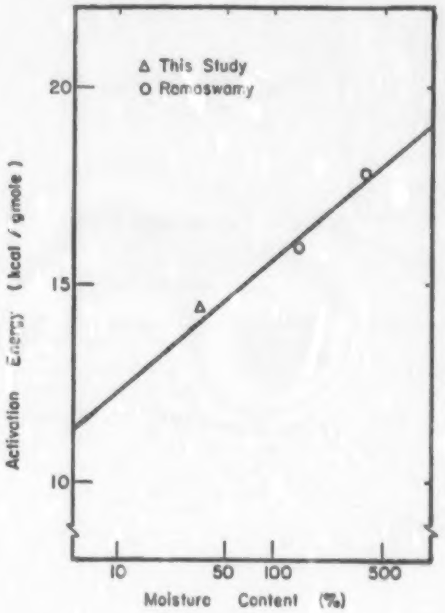


Figure 12
Effect of Moisture Content on the Activation Energy of Gas Production

E = Activated Energy
R = Gas constant
T_1, T_2 = Absolute temperatures.

At a moisture content of 400% and temperatures of 25°C and 35°C, the gas production rate data from Ramaswamy (22) result in an activation energy of 18,000 cal/mole, and 15,900 cal/g-mole, respectively. The data in the present study at a moisture content of 36% and temperatures of 17 and 26°C result in an activation energy of 14,500 cal/g-mole. The magnitude of these values indicates that the rate of gas production which parallels the hydrolysis of cellulose is chemically-rate controlled and not diffusion-rate limited (Figure 12).

CONCLUSIONS

The present study measured the rate and composition of gases released during anaerobic degradation of solid waste. The major gases observed were CO_2, H_2 and CH_4. The maximum rate of gas produced was 6.0 ℓ/kg dry weight during the 300-day testing period. It was further noted that increasing moisture contents increased the rate of gas production. A similar effect was caused by increasing temperatures and decreasing solid waste size. Increasing density of the solid waste tended to decrease the gas production. The H_2 was primarily detected in the containers with the 36% moisture content, while the CH_4 was found in the container with the unshredded refuse, possibly indicating that pH effect may exert an important effect on the initiation of methanic production phase.

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LEACHATE PRODUCTION AND VIRAL SURVIVAL FROM LANDFILLED MUNICIPAL SOLID WASTE*

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ABSTRACT

Results of five years of monitoring of a 435-ton batch-operated experimental landfill were presented. Leachate production, the major objective of the experiment, was emphasized. The concentration histories of selected contaminants, COD, Fe, Ca, Cl, selected metals, and indicator organisms were presented and discussed. Observed mass removals were compared to other values reported in the literature, and to the initial mass of solid waste disposed.

Work performed to achieve a secondary objective of the experimental landfill survival of viruses within the mass of landfilled solid waste, was also reported. Samples of shredded municipal waste, inoculated with poliovirus, were placed at three levels within the landfilled solid wastes. Laboratory studies were also performed to determine the survival of viruses and bacterial indicators of fecal contamination which were inoculated into leachate from the experimental landfill.

*Manuscript of the paper not received in time for publication.

DESIGN CRITERIA FOR GAS MIGRATION CONTROL DEVICES

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ABSTRACT

Principles of landfill gas migration under total and partial pressure gradients are discussed. Analytical principles relating to the design of control devices are presented, and concepts for field implementation of vent, barrier and hybrid systems are delineated.

Computer models for simulating gas migration both with and without control devices are described. Finally, several alternative protective installations for a typical landfill are evaluated and compared.

INTRODUCTION

Migration of methane gas around sanitary landfills constitutes a hazard in that the gas may accumulate to explosive concentrations in buildings adjacent to the landfill. The gas forms an explosive mixture in the range of 5 percent to 15 percent methane in air.

Previous research at the Ohio State University(1) has provided analytical technique for predicting the time-concentration profiles for methane around landfills. Computer programs(2) are available for studying site specific situations, and design charts(3) based upon simplifying assumptions may be used to estimate distances of migration for planning purposes.

Current research efforts are directed toward developing design criteria for installations to control methane migration. This paper describes the scope of the research and presents findings to date.

TYPES OF CONTROL DEVICES

Figure 1 delineates several types of control devices. Major classes include:

1. trench vents - trenches are continuously cut around the landfill and filled with coarse gravel. Such vents may vent naturally to the atmosphere or may undergo forced convection by mechanical pumping into or out of the trench.

2. pipe vents - similar to trenches except that they are placed at intervals around the landfills. Normally some type of convective flow must be used if such pipe vents are to be effective.

3. barriers - constructed similarly to trench vents except that the trenches are filled with saturated compacted clays or other impervious liner materials.

4. hybrid systems - a combination of trench vents backed by impervious barriers. The trench vent may or may not involve forced flow.

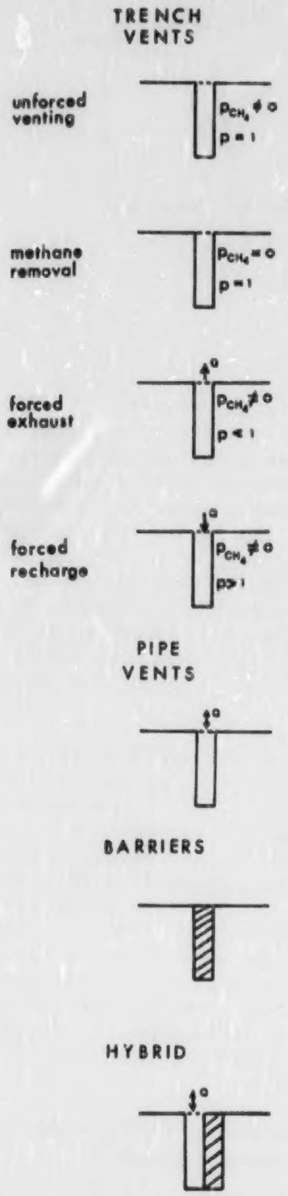

Figure 1 - Types of gas migration control devices.

The costs involved in the construction, maintenance and operation of these control devices vary widely, and it is important to optimize design to reduce cost and increase effectiveness.

COMPUTER PROGRAMS FOR OPTIMIZATION

Because of the many variables involved in the design of a gas migration control system, it is useful to be able to simulate the effectiveness of alternative design configurations on a computer. This approach allows for economically evaluating alternative solutions in order to select the optimum configuration. Such an approach also allows the designer to determine which design aspects most influence the effectiveness of the system so that important aspects can be carefully controlled during field installation. Finally, if the installed system does not perform as efficiently as desired, the computer simulation can be used to determine what modifications would most economically remedy the problem.

It is not the purpose of this paper to describe the details of the computer simulations. Therefore, suffice it to say that the several codes developed include treatment of:

1. soils displaying a distribution of pore sizes,
2. combined pressure and diffusional transition region flow,
3. bicomponent or multicomponent gas systems, and
4. two or three dimensional problems

Computer costs on the IBM 370/168 run from \$25 to \$2000 depending upon the complexity of the particular situation; however, many practical design problems cost of the order of \$100.

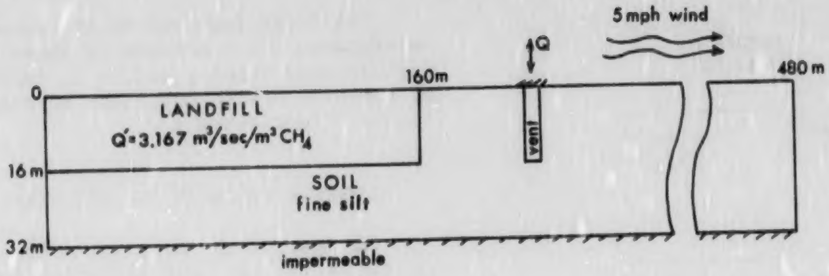

Figure 2 - Landfill configuration chosen for design problem.

TYPICAL DESIGN PROBLEM

In order to demonstrate some aspects of design of gas migration control devices, a specific case was developed. The configuration chosen is shown in figure 2. The landfill is circular in plan and of radius, 160 meters. The landfill is 16 meters deep. Impervious bedrock (or a groundwater table) is encountered at a depth of 32 meters. The soil beneath and surrounding the landfill is a fine dry silt having a porosity of 0.7. Venting at the ground surface is that which should be achieved by a uniform 5 mile per hour wind. In order to limit computer storage, a pure air boundary was imposed at a radius of 480 meters. The methane generation rate within the landfill was taken to be 3.167 cubic meters/second/cubic meter of refuse. For a three component case carbon dioxide was generated in the landfill at a rate of 1.357 cubic meters/second/cubic meter of refuse. This configuration was chosen for two reasons:

1. it represents a realistic situation, and

2. it poses a methane migration hazard in that the 5 percent methane contour migrates approximately 150 meters (450 feet) beyond the edge of the landfill.

Figure 3 shows contours of methane and carbon dioxide concentrations around the landfill when steady state is reached (approximately 20 years). The top two drawings give the results from a computer run simulating the simultaneous transport of carbon dioxide and methane into air while the lower drawing gives the results from a computer run simulating the transport of methane alone into air. The three component run shows slightly greater methane migration. However, because the difference is small and because the two component program is more economical to run, all results shown in this paper are for the two component configuration.

The remedial measures to be considered in this paper are based upon the assumption that the landfill was constructed without gas migration control devices. At the end of ten years, the relatively high methane concentrations surrounding the landfill resulted in the decision to install a gas migration control facility. The design constraints on the facility were that it was to be of the venting trench type, was to be located 32 meters from the edge of the landfill, and was to be equal in depth to the base of the landfill (16 meters). The questions raised are:

1. Is such a facility capable of controlling methane migration?

2. Will it be necessary and/or advantageous to use forced convection in the trench?

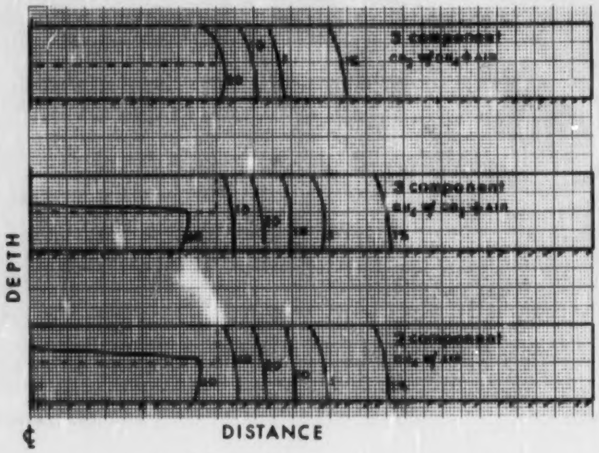

Figure 3 - Methane and carbon dioxide concentrations for two and three component versions of computer program.

DESIGN PROCEDURE

The first step in the design procedure is to determine if the basic configuration selected is capable of providing the desired controls. In order to do this, a computer simulation was performed for a perfect unpumped trench vent. Such a trench vent is one in which the total pressure remains atmospheric but from which all methane is instantaneously removed by an undisclosed means. Of course, such a situation cannot be realized in the field; however, it does represent a limiting condition which could be approached in practice.

The results of this computer simulation are shown in figure 4 as a plot of reduction ratio versus distance away from the landfill. The reduction ratio, to be used in several subsequent figures, is the ratio of the methane concentration at a given point and at a given time with a control device operating divided by the methane concentration at that same point and at that same time without the control device operating. Reduction ratio values have the following significance:

1. A reduction ratio of 1.0 implies that the control device in fact has no effect on the methane concentration.

2. A reduction ratio greater than 1.0 implies that the "control" device actually causes an increase in methane concentration.

3. A reduction ratio less than 1.0 implies that the control device is effective in reducing methane concentrations at that point and at that time. The lower the reduction ratio, the more effective the device.

In all of the figures in this paper, the reduction ratio is plotted when steady state has been attained after the control device begins to operate.

The distance scale is the ratio of the distance to a given point divided by the radius of the landfill (160 meters). The data shown on the figure are for points at the elevation of the base of the landfill.

Reference to figure 4 shows that the hypothetical control device which provides for complete removal of all methane instantaneously is indeed effective in reducing methane concentrations. Effectiveness is marginal between the landfill and the trench; however, significant reductions are achieved beyond the trench vent.

Given that it is possible to control methane migration using the configuration

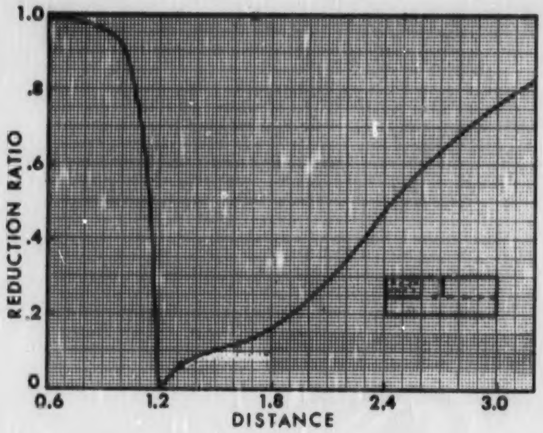

Figure 4 - Methane reduction ratios for perfect methane removal

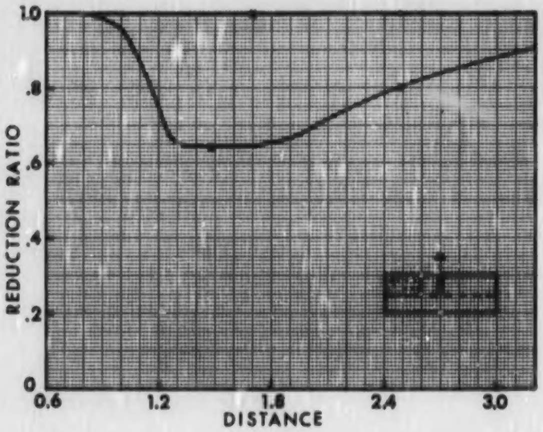

Figure 5 - Methane reduction ratios for unforced venting.

proposed, it is now necessary to determine how this configuration is to be implemented in the field. The first logical choice is to simply dig the trench, backfill it with gravel and allow unforced venting to occur. This situation is simulated by setting the total pressure within the trench to one atmosphere and increasing the efficiency of venting at the surface. Figure 5 shows the reduction ratio for this case. It is seen that significant reduction of methane concentration is realized. This design alternative could be considered to be practical.

The next alternative considered is forced exhaust. The system consists of fans installed every 15-24 meters (50 feet) along the trench and having a perfect seal on the top of the trench. The fans remove mixed gas from the system at a rate of Q cubic feet per minute (cfm). Figure 6 shows the methane reduction ratio for values of Q of 30,150 and 300 cfm. It may be seen that the methane concentrations increase between the landfill and the trench and decrease beyond the trench. For Q=150 cfm the 5 percent methane level is retracted to a distance of 69 meters. This design would probably be cosidered to be effective in controlling methane migration. However, it is important to note the possible affects of the higher methane concentrations between the landfill and the trench. The higher concentrations exist because the exhaust pumps actually attract methane from the landfill. Now, if the pumping system were shut down either intentionally or accidentally, the high methane concentrations between the landfill and the trench would provide a high diffusional driving force for transport of methane beyond the trench. Thus the consequences of system shutdown could be significant.

The next alternative considered is identical to the forced ventilation system shown except that air is pumped into the trench rather than out of it. Figure 7 shows the reduction ratio for Q=30,150 and 300 cfm. Significant reductions are found both between the landfill and the trench and beyond the trench. The reductions are greater for a given Q than those achieved by the exhaust venting system. Moreover, the possible detrimental effects of system shut-down are lessened since there is no methane buildup between the landfill and the trench.

Figure 8 shows a comparison of the several systems considered and for Q=300 cfm. Finally, figure 9 plots methane concentration contours for the several systems at steady state.

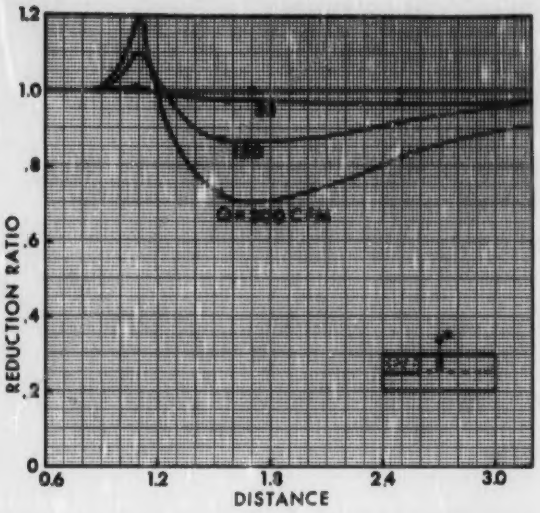

Figure 6 - Methane reduction ratios for pumped exhaust system.

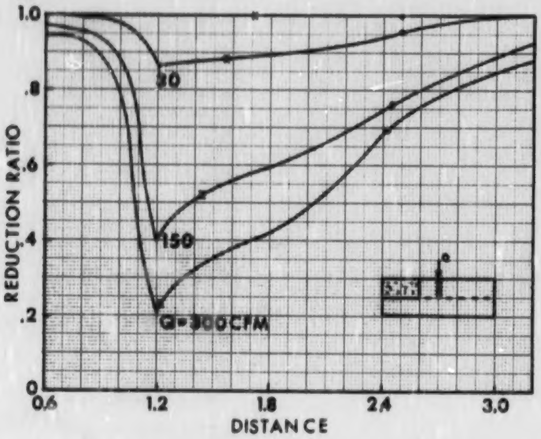

Figure 7 - Methane reduction ratios for pumped recharge system.

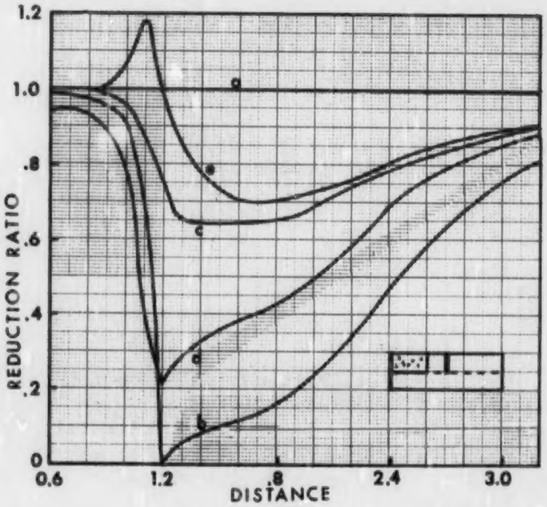

Figure 8 - Summary of methane reduction reduction ratios for the several design alternatives.
a. no control device
b. perfect methane removal
c. natural venting
d. forced recharge venting
e. forced exhaust venting

SUMMARY AND CONCLUSIONS

This paper has presented a typical design situation involving migration of methane to an unacceptably great distance from a sanitary landfill. Control measures involving both unforced and forced venting trenches were evaluated by computer simulation. The unforced trench configuration was effective. The forced exhaust trench configuration also proved effective; however, shutdown of the system could result in relatively rapid increases in methane concentration beyond the trench. The forced recharge configuration was the most effective system studied and did not pose as great a threat of methane buildup after system shutdown as did the forced exhaust configuration.

The reader is cautioned against interpreting the conclusions of this design example too broadly. In particular, the type of gas migration control devices which proved most satisfactory in the present example may or may not prove so for other cases. It is important that gas migration control devices for a particular site be designed in a site-specific manner.

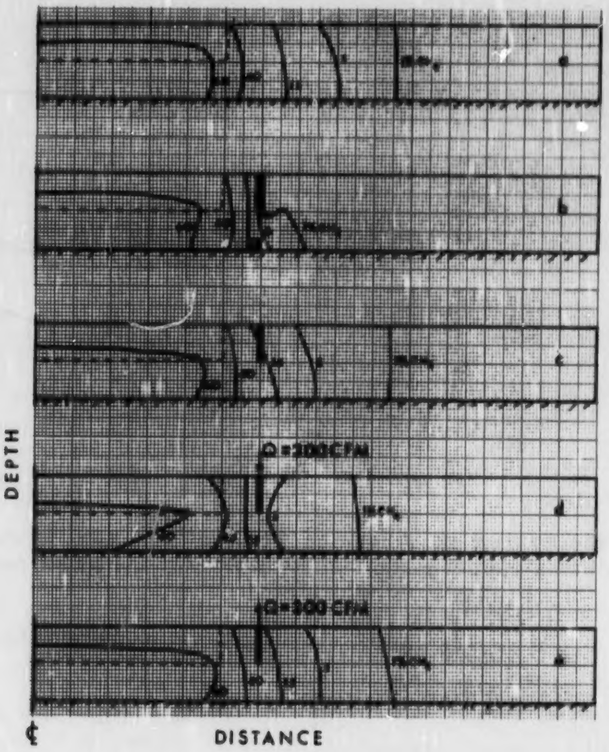

Figure 9 - Summary of steady state methane concentrations for the several design alternatives considered.
a. no control device
b. perfect methane removal
c. natural venting
d. forced recharge venting
e. forced exhaust venting

ACKNOWLEDGMENT

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MODELING OF LEACHATE AND SOIL INTERACTIONS IN AN AQUIFER

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ABSTRACT

The movement of leachate emanating from a hypothetical landfill is described by the two-dimensional convective-dispersive equation. Because the mass-averaged velocity of the convecting fluid appears as a coefficient in the transport equation, it is also necessary to solve the partial differential equations governing saturated-unsaturated fluid flow. Solutions of the resulting coupled set of equations for a hypothetical landfill located adjacent to a river provides considerable insight into the subsurface movement of landfill leachates.

INTRODUCTION

The effective design of a waste-disposal site requires an a priori understanding of the movement of landfill generated leachates into the subsurface. This movement is dependent upon the geologic and hydrologic character of the site as well as the physical and chemical interactions between the leachate and the soil. To forecast the movement of dissolved materials from the landfill one must depend either upon existing records of migration or upon models of the system. Because leachates generally move very slowly and data collection and analysis programs are both time consuming and expensive, records of migration are usually not available. Consequently simulation models, particulary mathematical models, can be useful tools, not only to obtain information necessary for rational landfill design, but also to formulate remedial schemes when unacceptable groundwater contamination has already occurred.

THEORETICAL DEVELOPMENT

A mathematical model is formally the solution of a set of equations which describe the physics underlying the movement of a convecting fluid and its dissolved chemical constituents. For systems of the complexity encountered in landfill simulation, these solutions are generally obtained numerically, although for some simplified situations analytical expressions do exist (Larson and Reeves, 1976).

The governing equations are based upon the conservation principle (mass, momentum and energy), augmented by constitutive relationships and proper initial and boundary conditions. After combination and simplification the governing equations reduce to a coupled pair of nonlinear, second-order partial differential equations; one descriptive of saturated-unsaturated fluid flow and the other characterizing the species transport. Once the governing equations are defined, their solution can be generated by a straightforward process of numerical manipulation. The main problems encountered during the simulation of non-conservative transport, therefore, are not so much a question of mathematical sophistication, as one of inadequate information on the chemical behavior of the dissolved ions in the system and estimation of the parameters entering the model.

Governing Equations

The equation governing saturated-unsaturated fluid flow in a vertical cross-section is given by

$$\frac{\partial}{\partial x}\left(k_x \frac{\partial h}{\partial x}\right) + \frac{\partial}{\partial z}\left(k_z \frac{\partial h}{\partial z} + k_z\right)$$

$$- \left(\frac{\theta}{n} S_s + C\right) \frac{\partial h}{\partial t} + Q = 0 \qquad (1)$$

where

$\underset{\approx}{k}=k_r\underset{\approx}{k}^s$ is the hydraulic conductivity $[LT^{-1}]$,

$k_r=k_r(h)$ is the relative hydraulic conductivity $[L^0]$,

$\underset{\approx}{k}^s$ is the hydraulic conductivity at saturation $[LT^{-1}]$,

θ is the volumetric moisture content $[L]$,

n is the porosity $[L^0]$,

S_s is the specific storage coefficient $[L^{-1}]$,

$C = n \frac{\partial S_w}{\partial h}$,

C is the soil moisture capacity $[L^{-1}]$,

S_w is the degree of fluid saturation $[L^0]$,

h is the pressure head $[L]$, and

Q is a fluid source (or sink) function $[T^{-1}]$.

To solve equation (1) additional information on the relationship between relative hydraulic conductivity and pressure head, and degree of saturation and pressure head is required. These functions are derived experimentally for each soil type encountered; typical curves for a coarse (sand) and fine (loam) soil are given in Figure 1.

The transport of dissolved ionic species in the leachate is described by the mass transport equation, which for a sorbing medium is written as

$$\frac{\partial}{\partial x}\left(\theta D_{xx} \frac{\partial c}{\partial x} + \theta D_{xz} \frac{\partial c}{\partial z} - q_x c\right)$$

$$+ \frac{\partial}{\partial z}\left(\theta D_{zx} \frac{\partial c}{\partial x} + \theta D_{zz} \frac{\partial c}{\partial z} - q_z c\right)$$

$$- \frac{\partial}{\partial t}(\theta c) - \rho \frac{\partial S}{\partial t} - \alpha \theta c + Qc^* = 0 \qquad (2)$$

where

c is the solute concentration $[ML^{-3}]$,

c^* is the concentration in the source (or sink) fluid $[ML^{-3}]$,

$\underset{\approx}{D}$ is the hydrodynamic dispersion coefficient $[L^2T^{-1}]$,

$\underset{\sim}{q}$ is the volumetric (phase averaged) fluid velocity $[LT^{-1}]$,

ρ is the (dry) bulk density of the soil $[ML^{-3}]$,

S is the adsorbed concentration $[M^0]$, and

α is the first order rate constant for decay $[T^{-1}]$.

Examination of Equation (2) reveals that the moisture content, θ, and the volumetric fluid velocity, $\underset{\sim}{q}$, are required before the concentration can be obtained. The moisture content can be read from Figure 1, given its value at saturation and the pressure head, and the velocity can be evaluated through Darcy's law:

$$q_x = - k_x \frac{\partial h}{\partial x}$$

$$q_z = - \left(k_z \frac{\partial h}{\partial z} + k_z\right) \qquad (3)$$

The dispersion coefficient, $\underset{\approx}{D}$, represents the effects of both molecular diffusion and mechanical dispersion. Scheidegger (1962) derived formulae for this coefficient, assuming a saturated system and isotropy of the medium with respect to the dispersivities. These formulae are assumed to hold also for unsaturated conditions, and given by

$$D_{xx} = D_L \frac{q_x q_x}{q^2} + D_T \frac{q_z q_z}{q^2} + D_d \tau \qquad (4a)$$

$$D_{zz} = D_T \frac{q_x q_x}{q^2} + D_L \frac{q_z q_z}{q^2} + D_d \tau \qquad (4b)$$

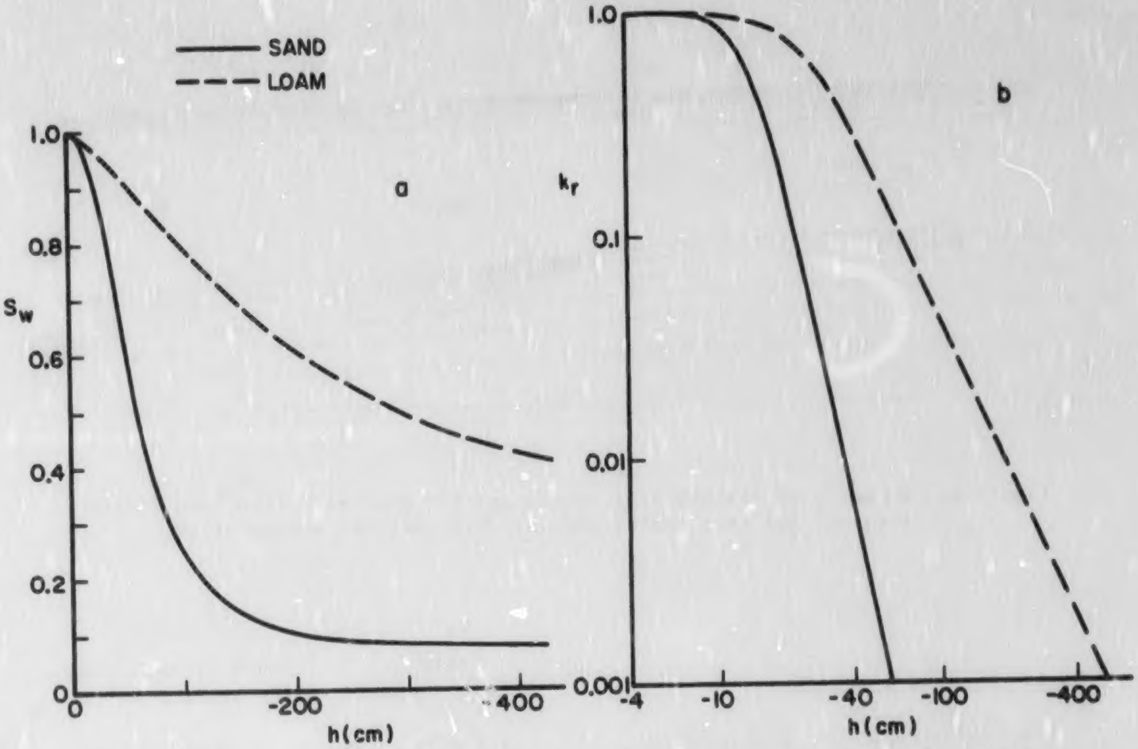

Figure 1. Degree of fluid saturation (a) and relative hydraulic conductivity (b) versus pressure head for the two soil materials used in the example transport problem.

$$D_{xz} = D_{zx} = (D_L - D_T)\frac{q_x q_z}{q^2} \qquad (4c)$$

where D_L and D_T are effective longitudinal and transversal dispersion coefficients $[L^2T^{-1}]$, which are related to the fluid velocity through

$$D_L = \lambda_L|q|/\theta \qquad D_T = \lambda_T|q|/\theta \qquad (5)$$

with the constants λ_L and λ_T representing the medium dispersivities [L], and where D_d is the molecular diffusion coefficient $[L^2T^{-1}]$ and τ is a tortuosity factor.

Method of Solution

Because equations (1) and (2) are coupled in such a way that the fluid flow is independent of the species concentration, the equations may be solved sequentially. The pressure head is obtained from (1), and substituted into (3) to yield the volumetric velocity. This in turn, is combined with (2) and (4) and the moisture content to generate the concentration. A method of solution for these equations, which has proven to be both accurate and efficient, is based on the finite element method using cubic Hermitian basis functions. This approach has been extensively tested under a variety of physical problems (Pinder *et al*, 1976), and is described in detail in existing publications (van Genuchten *et al*, 1977). For the purpose of this presentation it is sufficient to state that the procedure involves the discretization of the physical system into finite elements (see Figure 2), which are delineated by geometric information at nodes located at the corners of each element. While the dependent variables are specifically evaluated at each node, information is also available at any other location within the domain through Hermite interpolation.

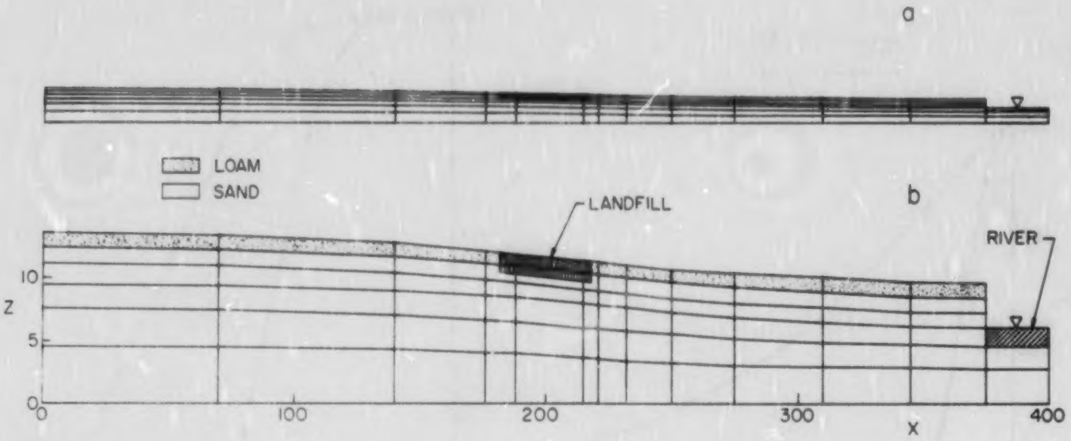

Figure 2. Geometry of the physical system and its discretization into finite elements (a) true scale, and (b) 5:1 vertical exaggeration.

APPLICATION

To demonstrate the utility of the model in evaluating the impact of a landfill on groundwater quality, a landfill located in a typical hydrological environment is considered. The physical system is illustrated in discretized form in Figure 2. The domain is bounded below by an impermeable (clay) layer, and symmetrically on each side by two small rivers, assumed to be 800 m apart. Because of the hydrological symmetry of the area only one half of the problem is considered. The cross-section consists of a 1-1½ m thick loam soil horizon overlying a sand aquifer. The following functions are used to characterize the hydraulic properties of the two soil materials (see also Figure 1),

$$S_w = \frac{\theta_r}{\theta_s} + \left(1 - \frac{\theta_r}{\theta_s}\right) \{1 + (\beta|h|)^{\gamma}\}^{-1} \qquad (6a)$$

$$k_r = \{1 + (a|h|)^{b}\}^{-r} \qquad (6b)$$

where θr and θs represent the residual and saturation moisture content of the soil, respectively. Values for the various constants β, γ, a, b and r in equation (6) are given in Table 1.

Table 1. Soil physical data used in the example transport problem.

variable	units	sand	loam
k_x^s, k_z^s	cm/day	300.	90.
n, θ_s	cm^3/cm^3	0.45	0.50
θ_r	cm^3/cm^3	0.031	0.10
β	cm^{-1}	0.0174	0.00481
γ	-	2.5	1.5
a	cm^{-1}	0.0667	0.04
b	-	5.0	3.5
r	-	1.0	0.64
S_s	cm^{-1}	0.00022	0.00020
λ_L	cm	60.0	60.0
λ_T	cm	40.0	40.0
τ	-	0.67	0.67

Precipitation and evaporation data, typical for eastern Long Island, New York, were used to calculate the net influx of water at the soil surface. Figure 3 shows the eight-year averaged distributions of monthly rainfall (R) and potential evaporation (E_p) which were used in the calculations. In order to obtain reasonable estimates of the actual evapotranspiration (E_a) the following reduction to E_p was applied

$$E_a = \begin{cases} E_p & R \geq E_p \\ R + 0.9\,(E_p - R) & R < E_p \end{cases}$$

Hence the net flux at the soil surface is

$$R_n = \begin{cases} R - E_p & R \geq E_p \\ 0.9\,(R - E_p) & R < E_p \end{cases}$$

The impact of the variable rainfall over the year is clearly illustrated in Figure 4 wherein calculated levels of the water table are presented. Simulation was started on an arbitrary date, January 1, 1968 (t = -2 years), assuming an arbitrary initial condition for the pressure head, h. The landfill, located

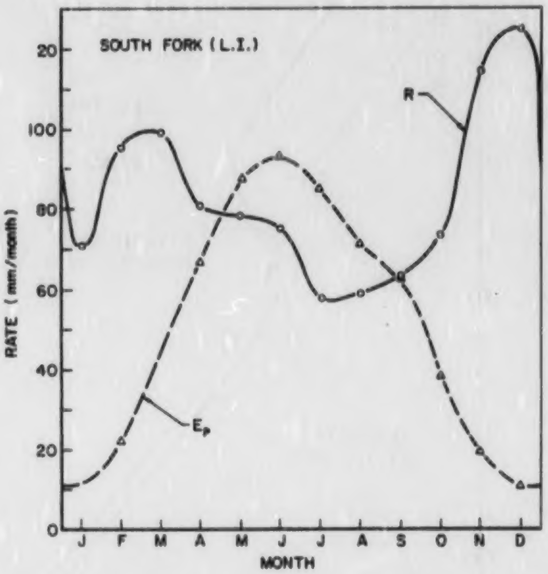

Figure 3. Patterns of precipitation (R) and potential evapotranspiration (E_p) employed in the transport problem.

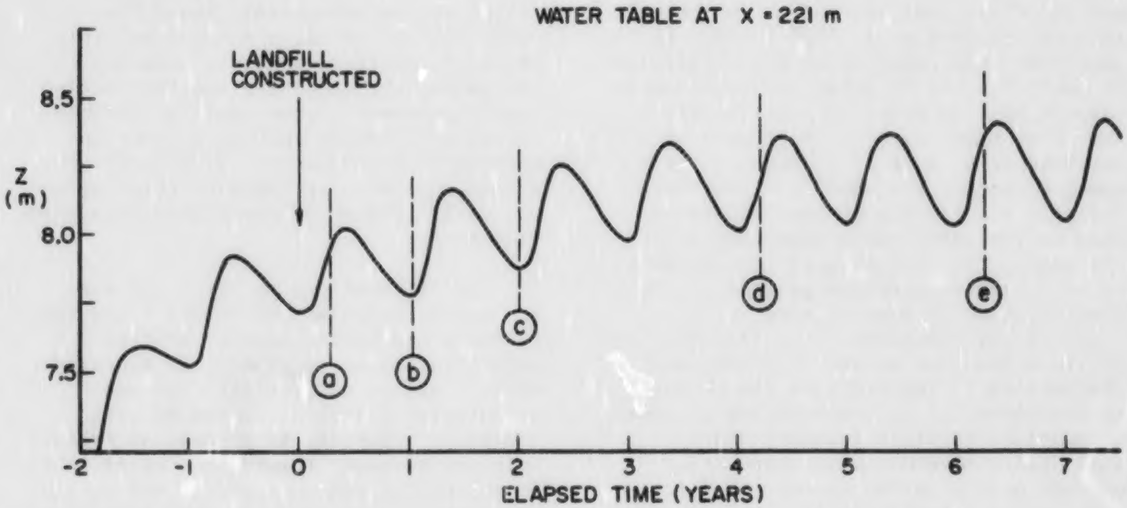

Figure 4. Calculated water table elevations along the vertical at x = 221m. The vertical dashed lines represent elapsed times at which calculated concentrations are presented.

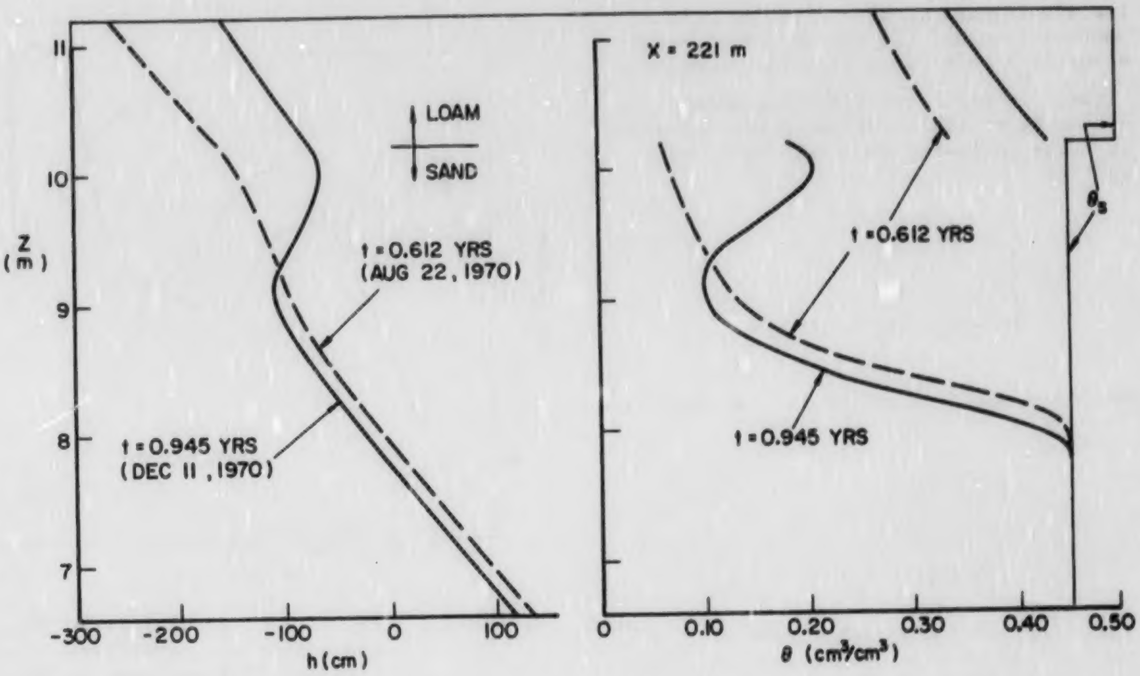

Figure 5. Calculated pressure head and moisture content profiles along the vertical at x = 221m.

halfway between the groundwater divide and the river, was assumed to be constructed on January 1, 1970 (t=0). At that time, the water table was stabilized at approximately 8m above the impermeable layer (about 3m below the soil surface). The water table data in the figure were obtained at a position 221m, immediately downgradient of the landfill. The data indicate a variation of approximately 35cm in the water table elevation during the year. The influence of the variable rainfall and evaporation pattern at the same location is demonstrated in Figure 5 by calculated profiles of pressure head and moisture content along the vertical. The profiles show a downward movement of moisture during periods of abundant rainfall (winter) and a slow upward movement when evaporation exceeds precipitation (summer). The water table responds to this as expected: rising in the winter and early spring and falling near the end of the summer. It is interesting to note that some horizontal movement of moisture towards the river takes place in the capillary fringe region immediately above the water table. In the present case, this region is confined to about 10cm due to the sandy make-up of the aquifer. Horizontal movement in the capillary fringe region will become much more important when finer soil material is present. For a loam aquifer, for example, this region may easily extend to about 30cm (see also Figure 1).

It is important to point out that the water table data of Figure 4 and the pressure head distributions of Figure 5 were obtained using eight-year averaged monthly values of precipitation and evaporation. The calculated results therefore represent only seasonal trends: moisture movement towards the water table in the winter and upward movement during the summer. The pressure head distributions will undoubtedly become much more erratic when observed hourly (or daily) rain and evaporation data are used. In that case, the direction of flow, especially in the upper part of the unsaturated zone, will be reversed several times during the day (or week). Such oscillatory behavior will be damped out, however, in the deeper layers of the profile, and it is unlikely that it will affect to any significant degree the transients of the saturated zone where most of the leachate movement will take place.

In this example, the landfill is located midway between the groundwater divide and the river. It is assumed that leachate emanating from the landfill is attributable to the downward movement of net rainfall and dissolution of landfill contents. The concentration of the solute in question is further assumed to remain at an arbitrary 100 mg/l level within the landfill at all times. No adsorption or decay processes are considered. The transport of the solute is illustrated in Figure 6. Examination of this figure reveals that the leachate moves through the unsaturated zone to the water table and then proceeds downgradient until it discharges into the river. The concentration of the leachate appears to remain high immediately below the landfill. A sharp concentration drop, however, is apparent when the concentration front reaches the ground water table. Dilution of the leachate with the flowing groundwater, along with some dispersion, causes the concentration to drop to approximately 15 mg/l further downgradient from the landfill. The results in Figure 5 show that significant amounts of material are being discharged into the stream after six or seven years. It is further apparent that some solute is transported through the unsaturated zone, mainly in the capillary fringe region and the region of water table fluctuations. For the present case (i.e., a sandy aquifer), significant pollution in the unsaturated zone is, however, limited to about 50cm above the water table, depending upon the season of the year. This unsaturated region of contamination appears to become more significant further downgradient of the landfill. The continuous, nearly horizontal, influx of solute into the capillary fringe region and the slow upward movement during periods of high evaporation, including the dispersion effects, are responsible for this presence in the unsaturated zone.

In this context, it may be interesting to consider the values of the longitudinal and transversal dispersivities used in the calculations (60 and 40cm, respectively). These values are very high when measured against published data in most soil physics literature. Several one-dimensional experimental studies indeed indicate longitudinal dispersivities in the order of one (Wood and Davidson, 1975) or only a few cm (Kirda et al, 1973; Biggar and Nielsen, 1976). These studies, almost without exception, are concerned with the entrance, downward movement, and redistribution of a solute carrying pulse of water. The temporal dimension of this process is generally limited to only a few hours or days. Simulations for the present study are carried out over a period of several years, using monthly values of rain and evaporation which keep the maximum fluid velocities limited to only a few millimeters per day. In addition, the time-frame of the present model does not allow for the flux oscillations near the soil surface which occur over a day or between two consecutive days which exhibit different rainfall and evaporation intensities. Such an oscillatory mechanism would tend to disperse the solute, if present, in the unsaturated zone. The only way to include these *apparent* dispersion effects into the physical model is through an increase in the dispersivity. The adopted value of 60cm for λ_L is obviously a first estimate; additional numerical experimentation seem necessary to better define this parameter. (e.g., by comparing model results using increasingly finer time dimensions).

Contrary to most soil physics studies, groundwater quality studies have generally used much higher values for λ_L and λ_T, especially when areal simulation models are used. Typical values in the literature range from a few meters (Pinder, 1973) to a seemingly unrealistic value of five miles (Amend et al, 1976). The argument for such high dispersivities is the fact that small scale heterogeneities in an aquifer cannot be detected by large-scale models. For example, when the convecting fluid encounters a small clay lens, the stream

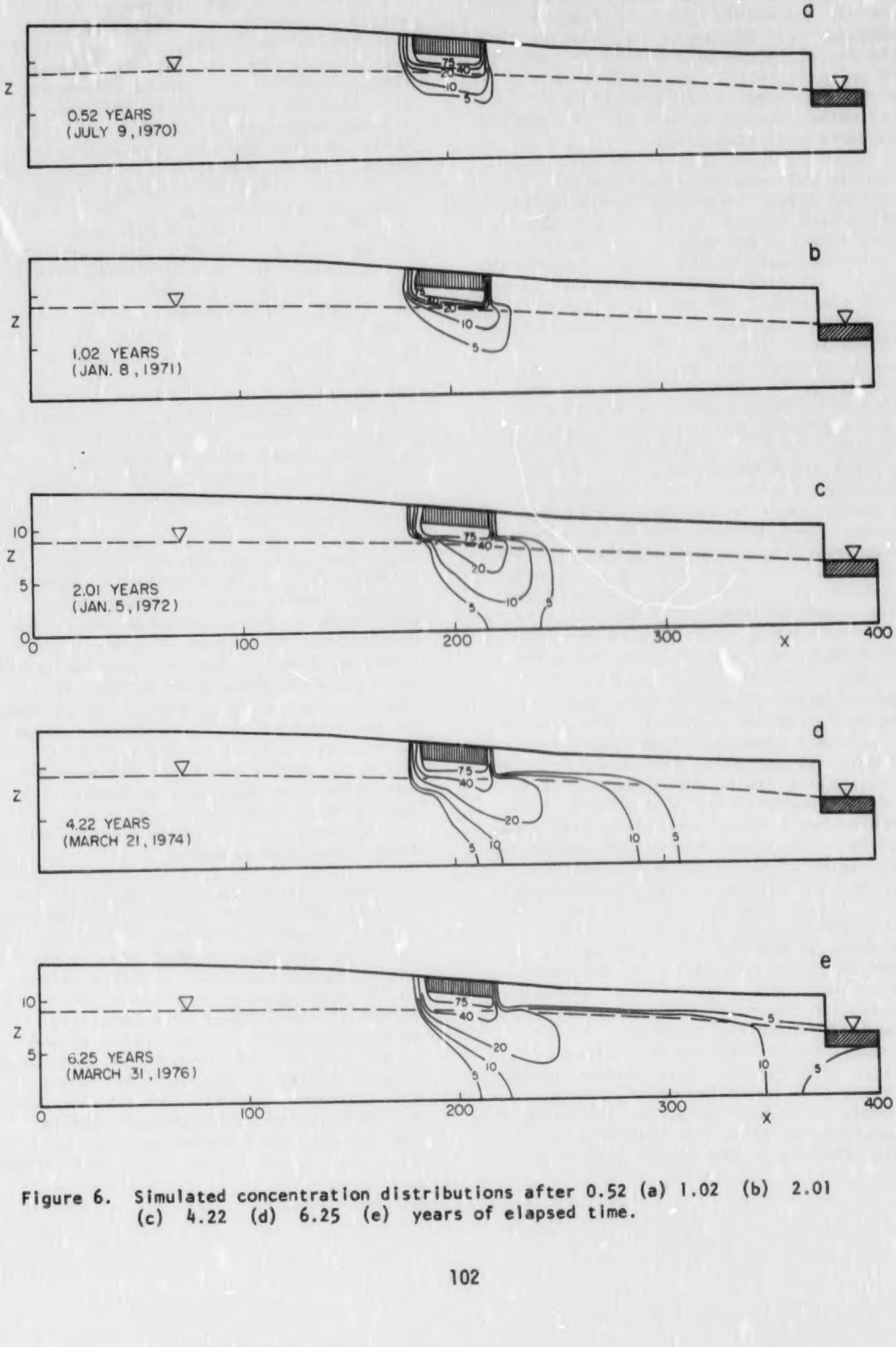
a
Z
0.52 YEARS
(JULY 9, 1970)
75
40
20
10
5
b
Z
1.02 YEARS
(JAN. 8, 1971)
75
40
20
10
5
c
10
Z
5
0
2.01 YEARS
(JAN. 5, 1972)
75
40
20
10
5
5
0
100
200
300
400
X
d
Z
4.22 YEARS
(MARCH 21, 1974)
75
40
20
5
10
10
5
e
10
Z
5
6.25 YEARS
(MARCH 31, 1976)
75
40
20
10
5
5
10
5
0
100
200
300
400
X

Figure 6. Simulated concentration distributions after 0.52 (a) 1.02 (b) 2.01 (c) 4.22 (d) 6.25 (e) years of elapsed time.

lines are diverted leading to an apparent dispersion of the solute. Because the macroscopic model is unable to include each clay lens, the above apparent dispersion must be lumped into the dispersivity. The need for a large value for dispersivity diminishes, however, when the physical system becomes better defined.

In the present example, the landfill was located midway between the groundwater divide and the river. A second example was also considered wherein the landfill was constructed above the divide itself. Results for this case indicated little movement away from the landfill after ten years of simulation due to the nearly horizontal groundwater table under the divide. It was apparent from this case that pollution was confined to a small region in the immediate vicinity of the landfill, although the solute concentrations in the groundwater reached a level of approximately 100 mg/l, i.e., much higher than the average concentration of 15 mg/l observed for the example problem discussed above. The different leaching patterns observed with the two examples clearly demonstrate the utility of mathematical simulation.

SUMMARY

The simulation of landfill leachate migration using numerical models provides considerable insight into the physical system. The importance of fluid transport and hydrodynamic dispersion are easily discernible from the numerical results. The example transport problem discussed in this paper further demonstrates the influence of the hydrological regime on the leaching pattern. While the present example focused upon the large scale picture, demonstrating its utility as a tool for landfill location, this type of simulation could also be used to advantage in establishing an optimal strategy in the landfill design itself.

ACKNOWLEDGMENT

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AERIAL DETECTION TECHNIQUES FOR LANDFILL POLLUTANTS

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ABSTRACT

A methodology for using remote sensing to detect landfill leachate contamination of ground and surface water is described. The problem is addressed without regard to specific geographic or climatological regions.

Among the topics covered are leachate indicators, spatial and temporal aspects of leachate detection, sensor selection, flight design and data interpretation. Specific methodologies for using remote sensing to detect leachate under various situations are outlined. These range from survey monitoring of individual landfills to comprehensive programs for regulatory monitoring of many landfills.

INTRODUCTION

One major disadvantage of landfill methods for solid waste disposal is the potential production of leachate, an extremely variable liquid resulting from waste decomposition and water flow through the waste. If not controlled, leachate can exit the landfill as part of, and as a contaminant of, the ground and surface water regime.

For planning remedial measures to control leachate from an existing landfill, all points of potential contamination must be located. This task is difficult, time-consuming and often inaccurate if conducted solely through field surveys; especially, since leachate breakout may occur at relatively long distances from the landfill. Although ground sampling and laboratory analysis are usually required to confirm leachate contamination, remote sensing techniques provide the most effective means for detecting potential locations of leachate breakout or contamination.

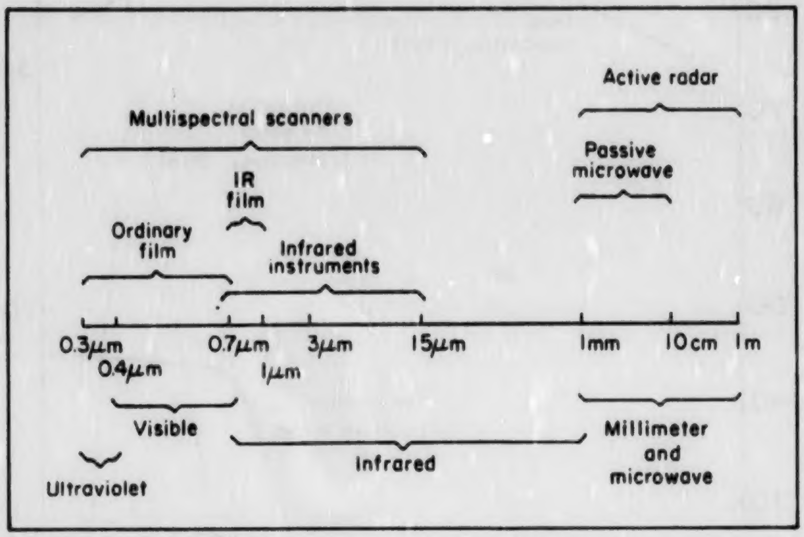
This paper provides a summary of a report, "Detecting Landfill Leachate Contamination Using Remote Sensors," which the authors are preparing for the U.S. Environmental Protection Agency (contract no. 68-03-2438). The work is an extension of an earlier study by Sangrey (12).

LEACHATE INDICATORS FOR SENSING

If leachate is to be detected directly or indirectly with remotely sensed data, then leachate or a leachate-related feature must, at the time of sensing, appear spectrally different from its surroundings or have some unique or identifiable spatial characteristic. The spectral and spatial indicators of leachate are listed in Table 1. They include observable features of leachate itself--wetness or an anomalous spectral response from water, soil, rock or snow--and observable effects of leachate--gaps in a vegetative or snow cover, or an anomalous spectral response from grass or taller vegetation. These indicators can be examined from the standpoint of detection with data acquired by airborne sensors of electromagnetic radiation (Figure 1).

ness or toxic substances can occur at longer distances from the landfill, 500 meters being quite reasonable.

Gaps in a vegetative or snow cover are usually easy to detect through sensing of reflected solar radiation in the visible and near-infrared spectral regions (Figure 2), of emitted radiation in the thermal infrared region, or of emitted or reflected radiation in the microwave region (11), (10) (8).



Scale variable

Figure 1. Electromagnetic Spectrum with Type of Sensors (after Holter, 1971)

Gaps

If large or small gaps in a vegetative or snow cover can be related spatially to a landfill, they often signal the presence of leachate. Caused by the leachate's wetness, toxicity or heat, the gaps can be isolated, or they can radiate from the landfill. Since heat would dissipate rather quickly with distance, heat-caused gaps should be found relatively close to the landfill. In contrast, gaps caused by wetness or toxic substances can occur at longer distances from the landfill, 500 meters being quite reasonable.

Since these gaps are notlimited to leachate- these gaps are not limited to leachate-affected sites, the primary task is to relate the gap to the landfill. Topographic and, to the extent possible, geologic analyses are normally required.

Wetness

Similar to gaps, any damp, saturated or puddled sites that can be related spatially to a landfill are potentially con-

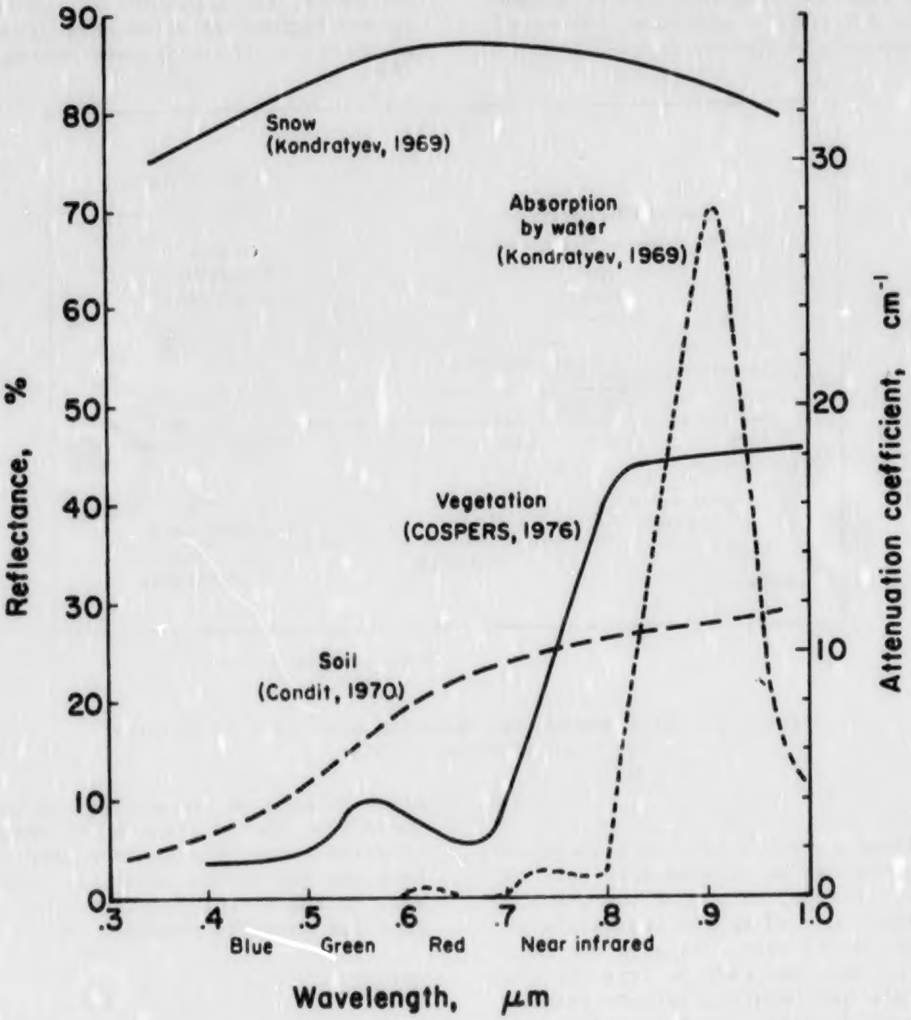


Figure 2: Spectral Reflectance of Typical Vegetation, Soil and Snow; and the Attenuation Coefficients of Water

taminated by leachate. As with gaps, wetness can radiate from the landfill or occur in small or large isolated spots, or seeps, the distance and direction from the landfill being highly variable.

Wetness can often be deduced from the vegetative types. If not, its spectral characteristics are sufficiently distinct that wetness is usually detectable over the entire electromagnetic spectrum (Bowers and Hanks, (2); Blanchard et al. (1); Ulaby et al. (15); Schmugge et al. (13). Although damp sites will not present as marked a contrast as saturated or puddled sites, at least the wetter sites will be separable.

Other Spectral Anomalies

Leachate will sometimes produce an anomalous spectral response besides those associated with wetness or gaps. Although these spectral anomalies usually provide a more positive indication of leachate than gaps or wetness, they must still be traceable to the landfill.

The spectrally reflective anomalies listed in Table 1 arise from possible differences in reflectance between leachate and soil, grass, snow or water, and between leachate-stressed and unstressed vegetation. Because of its high ferric iron content, for example, leachate may exhibit an unusually high red reflectance. Although this is not unique to leachate, the response contrasts sharply with that from grass, snow or otherwise clear water, and, to a lesser extent, with the response from soil or rock.

In a similar manner, lipid coatings on leachate-contaminated water should be detectable through passive or active sensing in the ultraviolet, blue, thermal or microwave regions, or at various Fraunhofer lines in the visible region (Kennedy and Wermund (6); Vizy (16); Watson et al. (17). The spectral reflectance of positively or negatively stressed taller vegetation might also show anomalies at any visible or near-infrared wavelength, though the near-infrared reflectance is apparently more sensitive to a broader range of stresses (COSPERS, 1976).

The spectrally emissive anomalies listed in Table 1 arise from the possible differences in temperatures and/or emissivities between leachate and soil, with or without grass; between leachate-affected and unaffected water; and between leachate-stressed and unstressed vegetation. Sensing for spectrally emissive anomalies can be performed in the thermal infrared or microwave regions (Figure 1).

TEMPORAL ASPECTS OF LEACHATE DETECTION

Temporal factors must be considered in a leachate detection program. This relates to the value of monitoring the development of the landfill from its initial stages to the present, and to the effects of season and time of day on the capacity to detect leachate.

For many landfills, the prevailing drainage conditions were established prior to the development of the site, and they have been changed little by the landfill operation. In other cases, the landfill operation has caused significant change at the site proper, but has not affected the subsurface drainage which surfaces at a distance from the site. Seldom will the development of the landfill alter the drainage so completely that it will be totally different from pre-landfill or, at least, early landfill conditions. Consequently, examination of remotely sensed images (e.g., aerial photographs) of the undeveloped and developing site will normally provide valuable information regarding where to expect leachate.

Considering seasonal effects, it should be obvious that since weather and climate are the major determinants of the amount of leachate produced, as well as the amount of vegetation or snow present to hinder detection, seasonal factors become especially important. In general, the potential for leachate production is high during wet periods and low during dry or

Table 1: Spectral and Spatial Indicators of Leachate

BACKGROUND/ SURROUNDINGS	GAP IN COVER*	WETNESS*	ANOMALOUS SPECTRAL RESPONSE REFLECTIVE	ANOMALOUS SPECTRAL RESPONSE EMISSIVE
soil/rock with grass cover	X	X	X	X
soil/rock with little or no grass cover		X	X	X
snow	X		X	
water			X	X
taller vegetation	X		X	X

*Although observable because of their spectral response, wetness and gaps in a vegetative or snow cover are listed separately for ease of discussion. Wetness ranges from damp areas to puddled water.

Table 2: Spectral Bands for Detecting Leachate Through Reflected Radiation

Leachate Indicator	Primary	Secondary
Gaps		
Vegetation/Soil, Rock	Infrared, Red	
Snow/Soil, Rock	Blue, Green	
Wetness		
Soil	Infrared	Red
Soil with Grass	Infrared	
Spectral Anomalies		
In Water	Red, Green	Blue
On Water (lipids)	Ultraviolet	Blue, Infrared
On Soil	Red, Green	Infrared
On Grass	Red	Infrared, Green
Stressed Vegetation	Infrared	Green, Red

freezing periods. The potential for leachate detection is normally lowest during times with deep snow or full canopies of taller vegetation.

Among other effects of season (and latitude) are the shadows which may obscure leachate indicators from overhead detection. As is obvious, sun shadows are also associated with the time of day. In most instances, the high sun angles at midday are best for detecting leachate indicators through reflected radiation. Although the water surface glint that accompanies higher sun angles may obscure an anomalous spectral response, it will, at least, facilitate the identification of wetness. In contrast, sensing of emitted radiation might best be performed during non-daylight hours to enhance thermal differences (Myers et al., (9).

RECOMMENDED PROCEDURES

Sensor Selection

The most useful wavelength intervals, or spectral bands, for detecting leachate indicators through reflected solar radiation are listed in Table 2. For remote detection of leachate, a sensor must provide data which allow an assessment of the spatial relationships between leachate indicators and the landfill under study. This requirement can only be filled effectively with an imaging sensor. Of the available imaging sensors, still or panoramic film cameras and scanning radiometers ("scanners") would be favored. The still camera might be a single or multiple lens frame camera, with any of several film-filter combinations. The scanner might be a thermal infrared or multispectral sensor (Table 3).

Although a multispectral scanner could be applied successfully in place of photographic camera systems, photographs are less expensive to acquire, process and analyze, and they are normally of higher spatial resolution. If photographic systems have the spectral capacity to monitor leachate in the ultraviolet, visible and near-infrared regions, the unique data acquirable by multispectral scanner are limited to the infrared region, particularly the thermal infrared. Since the thermal data will be most valuable if acquired during non-daylight hours, when other possible multispectral scanner data are not obtainable, a thermal infrared scanner would normally be preferred to a multispectral scanner with a thermal channel (Table 3).

Selection of a particular photographic system is dependent upon the number and types of landfills to be monitored, and the availability of equipment, facilities and/or funds. To illustrate, a local environmental group might wish to monitor a single landfill. This group would likely choose one photographic film for use with a hand-held, 35 mm camera. A color infrared film would be more generally applicable than other films (Tables 2 and 3). If monitoring an inundated landfill, where lipids might be expected, a black-and-white film filtered to receive ultraviolet and blue radiation should also be considered.

In contrast, a county environmental or health agency might employ one or more 70mm or 13cm format cameras, loaded with spectrally filtered, black-and-white films, or with some combination of color and black-and-white films; while a state monitoring agency might prefer the flexibility of a four-lens multiband camera, as outlined in Table 3. The U.S. Environmental Protection Agency, which has limited familiarity with the landfills and region to be overflown, might allow for all spectral and spatial indicators by carrying a 23cm format camera, loaded with color infrared film, and smaller format cameras, loaded with other films (including one imaging blue and ultraviolet radiation). Overall, many combinations are possible and effective.

Thermal sensing for leachate is considered an optional extension of the photographic program. In general, thermal data may: confirm or refute the interpre-

Table 3: Photographic Camera and Scanner Systems for Leachate Detection

SENSING OPTION	METHOD OF SENSING	BANDS SENSED AND RECORDED SEPARATELY	COMMENT
1. Photography			
a. color film	single camera; single image	B, G, and R* recorded as B, G, and R, respectively	UV can be sensed if recorded with B**; contrast of B layer will be lowered; proper exposure for UV and B will likely underexpose G & R
b. color infrared film	Idem	G, R, and IR recorded as B, G, and R respectively	UV and B cannot be sensed without affecting G, R, and IR
c. panchromatic film (black & white)	Multilens camera or several cameras, with spectral filters; multiple images	UV, B, G, and R, each recorded as black & white	Lower contrast of UV image will not affect other spectral images
d. black & white infrared film	Idem	UV, B, G, R, and IR, each recorded as black & white	Most multilens cameras have 4 lenses; lower contrast of UV image will not affect other spectral images
2. Multispectral scanner	Single scanner; magnetic tape, with or without image of one band off cathode ray tube or similar monitor	Any reflected or emitted bands from UV, visible & IR, including thermal; each band recorded as digital or analog signal on tape; if recorded in aircraft, one band as black & white film	Analog or digital data for any band or combination of bands can be printed on paper, displayed on video, or converted to photographic film
3. Thermal scanner	Single scanner; magnetic tape and/or image of one band off cathode ray tube or similar monitor	Commonly 8-14μm and/or 3-5μm; recorded as digital or analog signal on tape, or as black & white film	Idem, if recorded on tape

*B-Blue, G-Green, R-Red, IR-Infrared, UV-Ultraviolet
**Sensing of ultraviolet radiation will be limited by glass lens to wavelengths longer than about 0.36μm.

tation of a photo-identified indicator; provide some indication of the status of leachate contamination (e.g., if it is hot, a wet area is likely contaminated); detect other indicators which were overlooked or undetectable with photographic systems; or provide no additional information.

Flight Parameters

The design of an aircraft mission for detecting leachate is governed largely by the sensor(s) utilized and the spatial, spectral and temporal characteristics of leachate. The seasonal and diurnal characteristics of the indicators, as well as the local weather, set limits on the optimum time for sensing (Table 4).

Table 4: The Potential for Detecting Leachate under Different Vegetative and Seasonal Conditions

		VEGETATIVE COVER			
SEASONAL CONDITIONS	MODE OF SENSING	NONE	GRASS	PARTIAL CANOPY	FULL CANOPY
Wet	photo	E*	E	G-F	F-P
	thermal	E-G	E-G	F	P
Dry or Frozen	photo	E-G	G	F	P
	thermal	E	E	G	P
Partial or Light Snow	photo	F	F	F	P
	thermal	E	E	G	P
Full or Heavy Snow	photo	F-P	F-P	P	P
	thermal	G	G	G-F	P

*Ratings of Excellent (E), Good (G), Fair (F), and Poor (P) are subjective for midday photographic or pre-dawn thermal sensing. The two ratings are not equivalent.

For a given sensor, the size and/or spectral response of leachate indicators, and the size of the leachate-affected area, set limits on the flight height. For example, leachate-related gaps and wetness are quite variable in size; yet most leachate problems would be discovered if the photographic sensing could detect gaps and wetness of one meter across. If one-meter gaps and wet spots were spectrally distinct, they should be detectable with photographic scales of about 1:5,000 or larger. A 1:5,000 scale photograph could be obtained with a 150mm focal length camera by flying at 750 meters above ground, while shorter focal length cameras would require lower flying heights; H = 5,000 (f), where H = flight height in meters, and f = focal length in meters.

As regards flight parameters for thermal sensing, it is notable that the temperature of leachate near a landfill may be several degrees Celsius higher than uncontaminated waters in the vicinity. Thermal infrared scanners can detect apparent temperature differences of less than a degree (Reeves, 1975). While numerous factors must be considered in designing a thermal scanner mission, it is likely that the design flight height should be no more than 1,000 meters above ground for detecting a thermal leachate target. The mission should be conducted during pre-dawn hours as outlined in Table 4.

Data Analysis

The specific technique(s) applied in analyzing remotely sensed data for leachate indicators are dependent upon the form of remotely sensed data and the available equipment, time and/or funds for analysis. More sophisticated or costly analysis techniques are not synonymous with more information. Whatever the approach to analysis, the results must be applicable in the field; suspected sites of leachate must be located in the field, via maps or photographs, as well as in or on the analyzed data.

Several forms of data might be available for analysis: (1) panchromatic contact prints, (2) black-and-white transparencies of different visible and near-infrared spectral bands, (3) color and/or color infrared transparencies, (4) black-and-white transparencies of one or two thermal bands, and (5) magnetic tape containing digital or analog thermal data.

In general, the analysis may be visual, aided by magnification, stereoscopic viewing or filters, or the analysis may involve color enhancements through density slicing, additive-color viewing or subtractive-color viewing with diazo (Simonett, (14), Reeves (11). Densitometric measurements could provide quantitative values for selected features, but digitizing the image data is likely to be uneconomical and unwarranted.

DETECTION METHODOLOGY

The specific objective of a remote sensing program will determine the most appropriate detection methodology. Two cases can be defined. One is typical of regulatory monitoring where a general survey of a large number of waste disposal sites is conducted to evaluate conformance to regulations. The major elements in this type of methodology are:

Step 1. Obtain topographic maps which locate landfills to be monitored.

Step 2. Fly 1:5,000 scale, aerial photographic coverage of each landfill using film-filter combinations which record ultraviolet, blue, green, red, and near-infrared radiation.

Alternative A

Step 3A. Analyze photographs to identify the most probable locations of leachate breakout or contamination.

Step 4A. Field check these most probable locations.

Step 5A. Take appropriate action based on verification of leachate contamination.

Alternative B

Step 3B. Fly pre-dawn thermal infrared scanner coverage of all landfills.

Step 4B. Analyze thermal and photographic data.

Step 5B. (See Step 4A).

Step 6B. (See Step 5A).

Alternative C

Step 3C. (See Step 3A).

Step 4C. Based on the photographic analysis, select landfills to be overflown with pre-dawn thermal infrared scanner coverage. Conduct this sensing at a dry or frozen, low vegetation period to maximize effectiveness of this sensor.

Step 5C. (See Step 4A).

Step 6C. (See Step 5A).

The second general type of detection methodology applies to comprehensive control monitoring. The major elements in this type of monitoring are:

Step 1. Obtain all available background information on the landfill site, including topographic, soil and geologic maps and reports.

Step 2. Obtain aerial coverage of the undeveloped landfill site and coverage acquired periodically during the development of the site.

Step 3. Analyze available aerial coverage, together with background information, to identify the most probable locations of leachate breakout or contamination.

Step 4. Field check the landfill site(s), concentrating on those locations identified in Step 3.

Step 5. Fly new aerial photographic coverage of the landfills using one or more film-filter combinations which are appropriate for the expected spectral leachate indicators.

Step 6. Analyze the new photographs, together with the other aerial and background data, to identify the most probable locations of leachate breakout or contamination.

Step 7. Field check the landfill site(s).

Step 8. Upon verification of leachate contamination, plan remedial measures.

(Steps 9 through 12 are optional extensions)

Step 9. Fly pre-dawn, thermal infrared coverage of the landfills.

Step 10. Analyze the thermal infrared data, together will all photographic and background data, to identify any additional locations of suspected leachate breakout or contamination.

Step 11. Field check any new locations of suspected leachate.

Step 12. If required, modify planned remedial measures.

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Preliminary analysis of these data was done by W. L. Teng under the direction of Professor T. Liang from the School of Civil and Environmental Engineering at Cornell University.

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POLLUTANT MIGRATION PATTERNS FROM LANDFILLS

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ABSTRACT

A survey of three municipal landfill sites was conducted under the sponsorship of the U. S. Environmental Protection Agency. The objective of the survey was to identify contaminants and determine their distribution in the soil and groundwater beneath the landfill sites. Three sites were selected for the study and the sites represented varying geologic conditions, recharge rates, and age, ranging from a site that had been closed for 15 years to a site currently operating.

Site investigations included a preliminary geologic investigation to establish boring locations. Generally, seven to nine borings were made in the sites and were distributed up-gradient and down-gradient with respect to groundwater contours, and within the site. Soil cores were removed from a boring at varying locations depending on the distance from the bottoms of the landfill to groundwater. Undisturbed samples were obtained for soil properties determination and disturbed samples were obtained for chemical analysis. A groundwater sample was obtained from each boring location.

Soil properties examined included permeability, density, water content, and grain size analysis. Chemical analysis of the soil samples included a water and acid extract of soil samples. Filtered water extract samples were analyzed for 23 chemical parameters, and filtered acid extract samples were analyzed for 12 trace metals. Groundwater samples were analyzed for the 23 parameters used for the water leach.

Presentation of the results of the site survey will include the soil properties tests and an analysis of the chemical properties for general trends. Emphasis will be placed on comparing and contrasting the sites studied.

INTRODUCTION

Ultimate disposal of municipal solid wastes by landfilling accounts for the disposition of a majority of the solid waste produced in this country (1). In many areas of the country, net recharge of groundwater aquifers occurs and represents a potential pathway for contaminant migration from the solid waste to these aquifers. Because the net turnover time for groundwater aquifers is extremely long, as compared to surface water supplies, it is critical that these sources of water be protected from contamination via solid waste disposal.

Municipal solid waste disposal by sanitary landfills usually incorporates a number of design considerations to reduce the risk of contaminant migration. The most important of these include proper site selection, preparation, and disposal operation. In many cases, a minimum distance between the refuse and groundwater is required to take advantage of filtration, dispersion, and attenuation within the soil matrix if leaching does occur. Investigations in the past concerning contaminant migration from landfill have primarily concentrated on either monitoring programs examining the input to groundwater aquifers (2) or investigations of soils and attenuation associated with landfill sites (3). These studies seek to provide information that can be used to strengthen current design and operational guidelines, thus minimizing the threat of groundwater contamination from municipal solid waste disposal.

The study described herein is a survey of three landfill sites in the Eastern United States. The sites represented varying geologic conditions, recharge rates, age (ranging from a site closed for 15 years to a currently operating site), and distances between refuse and groundwater, ranging from 2-20 meters. The objective of the study was to characterize the soil and groundwater beneath and in the immediate vicinity of the landfill to identify and determine the distribution of contaminants associated with the landfill sites. Primary emphasis was placed on trace metals, the influence of soil extraction method for determining availability, and physical characteristics of soils from the respective site as they influence contaminant distribution.

METHODS

Site Selection

Sites for inclusion within the study were selected from a list of available sites considering the variables mentioned previously. Preliminary evaluation was based on geologic information available, including groundwater hydrology, and previous records or studies performed on the sites. The three sites selected represent a diverse group based on geologic conditions, age, and depth to groundwater.

Site Sampling

Based on preliminary information available, a sampling plan was prepared for each site. In general, this sampling plan included a combination of borings both within the site (internal) and around the site (external). External borings were arranged in such a manner that a portion of these borings were upgradient and a portion were downgradient with respect to groundwater flow at the site. A generalized boring location scheme is presented in Figure 1.

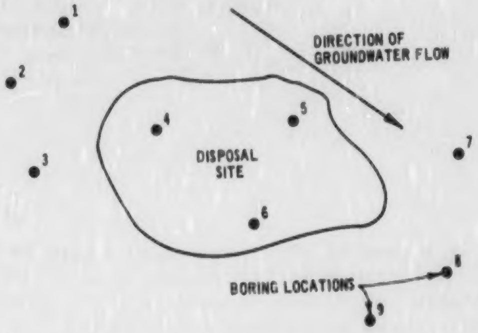

Figure 1. Generalized Sampling Plan.

Sampling from individual borings included sets of companion samples. These companion samples represented undisturbed samples for soil analysis and disturbed samples for chemical analysis. Borings were made with truck-mounted rotary drill using a hollow stem flight auger. Samples were taken with a Hvorslev fixed-piston sampler in a Shelby tube or a split-spoon sampler.

Vertical sampling within borings was arranged depending on the distance between the refuse and groundwater (zone of saturation). The first sample was taken at the refuse-soil interface and the last sample was taken at (or just above) the zone of saturation. Three to seven samples were taken at a particular boring dependent on the distance between the refuse and groundwater. A groundwater sample was obtained with a bailer 48 hours after the boring was made. A generalized vertical boring profile is presented in Figure 2. To maintain consistency between borings, samples were taken at approximately the same elevations between borings.

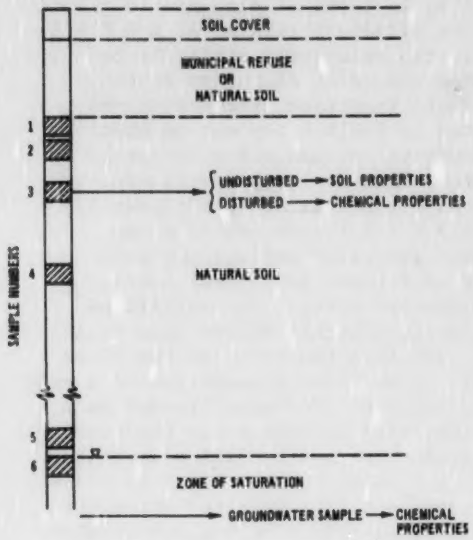
SOIL COVER
MUNICIPAL REFUSE OR NATURAL SOIL
1
2
3
UNDISTURBED → SOIL PROPERTIES
DISTURBED → CHEMICAL PROPERTIES
SAMPLE NUMBERS
4
NATURAL SOIL
5
6
ZONE OF SATURATION
GROUNDWATER SAMPLE → CHEMICAL PROPERTIES

Figure 2. Vertical Boring Profile.

Sample Analysis

Soil Properties. Undisturbed soil samples were analyzed for the following properties: (1) Grain size analysis, (2) Atterberg limits, (3) Dry density, (4) Wet density, and (5) Permeability. These properties allowed classification of individual samples by the USCS system (4).

Chemical Properties. Chemical analysis of soil samples were performed by making either a water or acid extraction of samples as received from the field. Moisture contents were determined on sub-samples to permit calculation of constituent concentrations on an oven-dry weight basis. Water extracts and processed groundwater samples were preserved by the appropriate methods (5). A list of the chemical species determined in the extracts or groundwater samples is presented in Table 1.

Water Extraction. Two hundred grams of soil of known moisture content were weighed into 1000-ml polycarbonate centrifuge bottles, 600 ml distilled-deionized water added, the bottles sealed, and then shaken on a rotary shaker for one hour. After shaking, the bottles were centrifuged at 2200 rpm for 30 minutes and the supernatant was filtered through a 0.45-μm millipore filter.

TABLE 1

LIST OF CHEMICAL PARAMETERS

CHEMICAL SPECIES	GROUNDWATER FILTRATE	EXTRACT	
		WATER	NITRIC ACID
SO_4	X	X	
SO_3	X	X	
Cl	X	X	
NO_3	X	X	
NO_2	X	X	
TOC	X	X	
Mg	X	X	
Ca	X	X	
Na	X	X	
CN	X	X	
As	X	X	X
Be	X	X	X
Cd	X	X	X
Cr	X	X	X
Cu	X	X	X
Mn	X	X	X
Pb	X	X	X
Zn	X	X	X
B	X	X	X
Se	X	X	X
Hg	X	X	X

Acid Extraction. Fifty grams of wet soil of known moisture content were weighed into 250-ml teflon beakers, and 60 ml of 8N nitric acid (HNO_3) were added. The soil-acid suspensions were heated to 95°C for 45 minutes and stirred every 15 minutes. After being allowed to cool to room temperature, the suspensions were transferred and filtered with 8N HNO_3 through a 0.45-μm filter. The digested soil was washed three times with 20-ml portions of 8N HNO_3. The filtrate was transferred to 250-ml volumetric flasks and diluted to volume with 8N HNO_3.

Groundwater Samples. Upon arrival at the laboratory, the groundwater samples were centrifuged at 2200 rpm for 30 minutes. The resulting supernatant was then filtered through a 0.45-μm millipore filter and preserved for chemical analysis.

Data Analysis

To facilitate subsequent data analysis, the results of chemical analysis were appropriately coded, and the sample coordinates were appended. Horizontal coordinates were calculated from a reference point at each site and the vertical coordinate was determined from the boring log.

Analysis of the resultant data was performed by use of multivariate statistical techniques. These techniques are particularly advantageous for this type of study since they seek to reduce the variable set to some interpretable grouping of variables. Relationships among the sites were determined by factor analysis (6) on the following groups of chemical analyses: (1) groundwater plus water extracts, (2) water extracts, and (3) acid extracts. Variables (i.e., chemical species) that demonstrated no variance for a particular site (e.g., samples were consistently below detection limits) were deleted from the data matrix prior to analysis. Relationships between sites were explored by means of discriminant analysis (7). As in the case of factor analysis, variables that possessed no variance between sites were eliminated from the data matrix prior to analysis.

The underlying principle behind both factor analysis and discriminant analysis is to construct factors that are linear combinations of the original data set. In the case of factor analysis, this is aimed at maximizing the variance of the original data accounted for by as few factors as possible. In the case of discriminant analysis, this is aimed at explaining the separation of data sets by as few factors as possible. In both types of analysis, the use is generally exploratory in nature; that is, attempting to reduce the number of variables required to explain behavior of the data.

RESULTS AND DISCUSSION

General Site Characteristics and Soil Properties

The results of tests on soil properties are presented as a range of values for the respective sites in Table 2. As is evident from this information, the three sets varied significantly with respect to geologic properties. This variation was expected to influence the results and interpretation of the chemical analysis.

The elevation data show the thickness of refuse and the relationship of the refuse to the groundwater. In terms of refuse thickness, the sites are related as $C > A \geq B$; in terms of distance to groundwater the sites are related as $A > C \geq B$. Since it was relatively difficult to determine the exact elevation of the refuse-soil interface, the elevations presented in Table 2 may not be exact but relationships between sites are probably accurate. USCS soil types exhibited by the three sites indicate gradations in which $A > C > B$, water contents and densities generally reflect the soil classification and geological setting of the respective sites. Permeabilities generally fell within values that would be expected from the soils at the sites although sites B and C demonstrated a wide range of values. Permeability and soil characteristics are extremely important in relationship to attenuation capacity of the respective sites. Based on these characteristics, the expected attenuation in the soil at the sites would be ranked as $B > C > A$. Little or no contaminant attenuation by the soil would be expected at Site A.

In addition to the geological and physical properties of the sites, an attempt was made to characterize the refuse at the various sites. Based on historical records available from each site surveyed, the age or the sites were ranked as $A > B > C$. In the case of site C, the facility was operating as a sanitary landfill at the time of the survey.

Chemical Characteristics

The procedures utilized for analyzing the chemical data from the samples were selected to condense the data and reduce the number of variables to be considered. The results of the factor analysis were displayed by plotting the factor scores obtained for the first two factors. The scores are computed to have a mean of 50.0 and standard deviation of 10.0 for each factor. The explained variance is noted for each plot and the major variables contributing to each factor are displayed along the respective axes. A similar procedure was used to display the results of the discriminant analysis conducted on the data from all sites. For purposes of clarity, the chemical data from the water extract samples plus groundwater and acid extract samples were treated separately.

TABLE 2

SITE GEOLOGY AND PHYSICAL PROPERTIES OF SOILS

SITE	GEOLOGICAL SETTING	ELEVATION meters REFUSE	 NATURAL SOIL	 GROUNDWATER	USCS SOIL TYPE	WATER CONTENT %	DENSITY gm/cm²	PERMEABILITY cm/sec
A	GLACIAL TERRACE	258-259	254-256	231-232	SP	2.4- 7.5	1.51-1.78	2.0×10^{-2} 1.0×10^{-1}
B	AEOLIAN LOESS	52- 67	49- 64	50- 61	ML,CL	20.1-33.3	1.40-1.57	8.0×10^{-3} 8.0×10^{-5}
C	RESIDUAL SILTS AND CLAYS	145-157	122-132	122-132	SM,ML	14.9-28.9	1.52-1.78	3.0×10^{-7} 3.0×10^{-4}

Water Extracts. The water extracts performed on the soil samples were intended to demonstrate: (1) if water soluble contaminants originating from the refuse had migrated into the natural soil (or groundwater), and (2) if water soluble contaminants had been removed from the soil underlying the landfill sites. In the case of the water extract samples factored for site B, no demonstrable differences could be detected within the sample set; therefore, this site was not considered under this section, except for the results of the discriminant analysis.

The results of factoring the water extracts and groundwater data for sites A and C are presented in Figures 3 and 4, respectively. For site A, there is a significant separation between groundwater and water extract samples. In the case of the first factor, Figure 3, all water extract samples appeared to possess higher levels of chloride and cyanide than the groundwater samples. Because there is little separation between water extract samples, separation for factor I represents background conditions. In contrast, there is significant separation between groundwater samples on factor II, representing magnesium and calcium. Furthermore, the points in the lower left quadrant represent up-gradient groundwater samples taken down-gradient and beneath the landfill. This would indicate that calcium and magnesium have been introduced into the aquifer as a result of leaching from the landfill. In the case of site C, a separation between several points representing groundwater samples and water extract samples within the zone of saturation is evident, Figure 4. Separation is significant for both factors obtained. The three points to the extreme right in Figure 4 represent samples obtained down-gradient from the

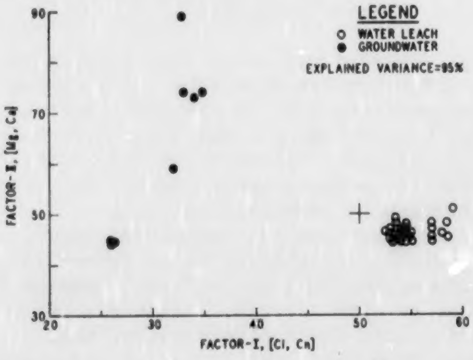

Figure 3. Factor Analysis, Site A: Water Extracts Plus Groundwater

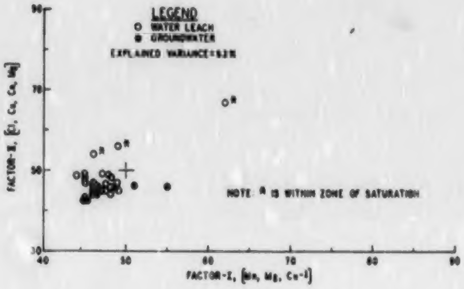

Figure 4. Factor Analysis, Site C: Water Extracts Plus Groundwater

landfill. Similar to site A, this indicates that certain contaminants have been introduced into the aquifer as a result of leaching.

The results of factoring the water extract samples for sites A and C are presented in Figures 5 and 6, respectively.

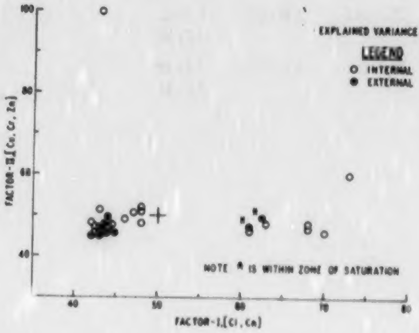

Figure 5. Factor Analysis, Site A: Water Extracts

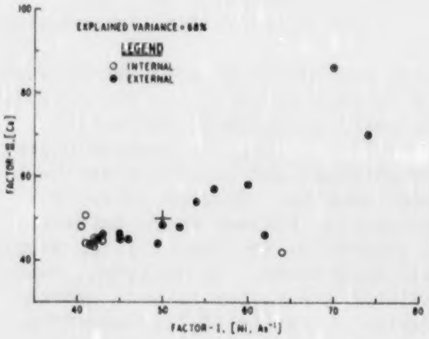

Figure 6. Factor Analysis, Site C: Water Extracts

For site A there exists some separation between internal and external samples on factor I, Figure 5. This factor is similar to factor I previously obtained for site A, Figure 3. This may indicate that chloride and cyanide are being introduced into the underlying soil by leaching. Due to the high permeability of soils beneath site A, this condition may be expected to occur. There is no significant separation of external and internal points on factor II except for one point that may be an outlier. In contrast there is significant separation between internal and external points for site C. Factor I is represented by nickel and arsenic (inverse relationship), and factor II is represented by calcium. These data indicate that arsenic has originated from the landfill, and apparent leaching of naturally occurring nickel and calcium from the soil. The leaching of calcium from underlying soils has been previously demonstrated and shown to be related to the migration of leachate from a landfill (8,9).

The data from the water extract samples for all sites were analyzed by discriminant analysis, and the results are presented in Figure 7. Similar to factor analysis, major variables contributing to separation between data sets are noted on the respective axes. A distinct separation exists between sites A and C and site B on the discriminant factor I, sodium. This separation is explained primarily by differences in chemistry and soil mineralogy between the respective sites. In contract, the separation on factor II is between internal and external samples as opposed to site location. In terms of arsenic, this implies movement of this element from the landfill to the underlying soil. In the case of the remaining elements (calcium, magnesium, nickel, and lead), an inverse relationship exists indicating the leaching of these elements from the underlying soils due to the presence of the landfill. Because of the age of two of the sites (i.e., A and B), this leaching may not be detected in groundwater samples taken at the time of

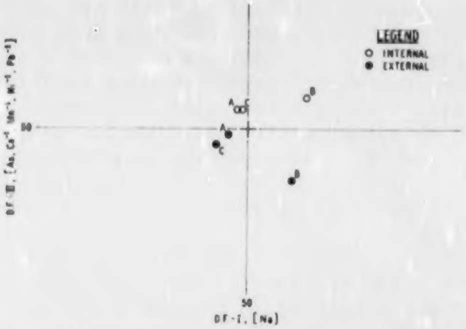

Figure 7. Discriminant Analysis, All Sites: Water Extracts

the site survey.

Acid Extracts. The acid extract performed on the samples was intended to dissolve or bring into solution all but certain mineral structures of the samples (10). In this manner, comparisons could be made between internal and external samples to determine the extent of attenuation that had occurred (11). The results of factoring and acid extract data for sites, A, B, and C are presented in Figures 8-10, respectively. In the case of site A. both internal and external points demonstrate a large variability. Because of the physical and geological characteristics of this site, this variability may be expected. No definite conclusion with respect to attenuation may be reached from this analysis of data on site A. For site B, some separation of external and internal sample points was evident, Figure 9, for both factors indicating that some degree of attenuation had occurred in the soil beneath the site. Because of the low permeability generally exhibited at site B, the separation of points (i.e., degree of attenuation) was not too significant with samples obtained from various depths beneath the landfill. The results of factoring for site C, Figure 10, does demonstrate a significant amount of attenuation as evidenced by the separation between internal and external points. Separation appears to be most significant for factor I, zinc and nickel. It is interesting to note that the evaluation of data for site C appears to contradict the evaluation based on the water extract samples. Examination of the factors related to nickel appear to indicate leaching in the case of the water extract samples but attenuation in the case of acid extract samples. This anomaly may possibly be explained by a conversion of water soluble nickel compounds in the soil under site C to more insoluble forms by the presence of the landfill.

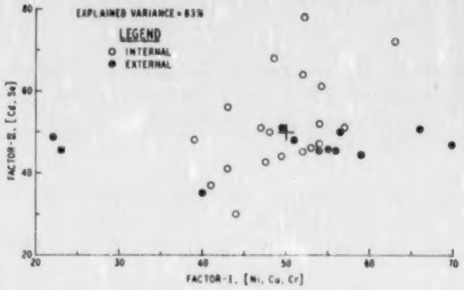

Figure 8. Factor Analysis, Site A: Acid Extracts

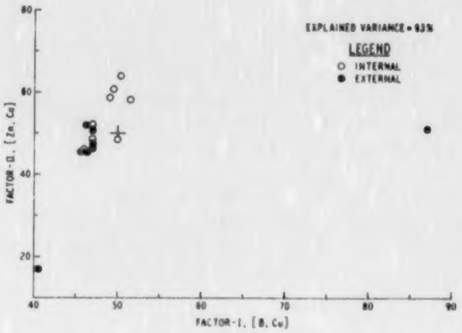

Figure 9. Factor Analysis, Site B: Acid Extracts

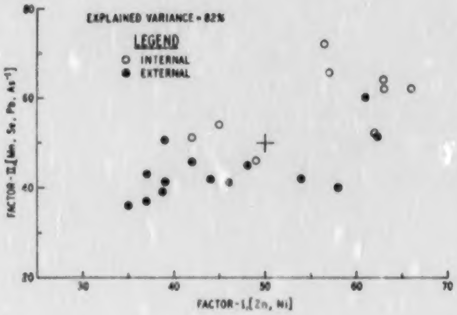

Figure 10. Factor Analysis, Site C: Acid Extracts

The results of the acid leach data are summarized for all sites through discriminant analysis and are presented in Figure 11. Evaluation is similar to the

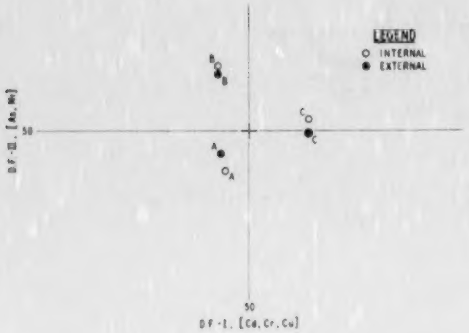

Figure 11. Discriminant Analysis, All Sites: Acid Extracts.

case for water leached samples performed previously. Separation on factor I appears to be solely dependent on the site selected, while separation on factor II is a function of sample location within a site. The presence of arsenic and its attenuation, alluded to earlier, are confirmed in Figure 11. The conversion of nickel to a more insoluble form is confirmed by a comparison of Figures 7 and 11. These conclusions seem to hold for all sites with the exception of A. In this case, the opposite effect appears to have occurred.

Summary of Chemical Properties

The leaching of contaminants from landfills into subsurface soils and aquifers occurs at municipal landfill sites. Additionally, the leachate may react with certain elements in the subsurface soil and cause their subsequent migration. These events are dependent on site conditions although evidence for both addition and removal of contaminants was found in this study. Certain contaminants, once mobilized through leaching, may become attenuated in the subsurface soil and continued leaching may convert some contaminants to forms that are water insoluble and more resistant to leaching.

The data analysis presented in this paper does have validity in interpreting and summarizing the results from this site survey. Since the derived factors in this study represent combinations of original variables, the behavior inferred may hold for groups of contaminants represented by the various factors. Application of these techniques to related studies may reveal information concerning the migration or behavior of various contaminants that may be useful in designing monitoring programs for municipal sanitary landfills.

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LAND DISPOSAL CRITERIA AND COMPLIANCE MONITORING RELATIVE TO LEACHATE AND GROUND WATER

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I am working on the solid waste disposal regulations to be promulgated by EPA under authority of Section 4004(a) of the Resource Conservation and Recovery Act of 1976 (PL 94-580). These regulations are to contain criteria for determining which solid waste disposal facilities shall be classified as sanitary landfills and which shall be classified as open dumps. Such criteria are to provide that a facility may be classified as a sanitary landfill and not an open dump only if there is no reasonable probability of adverse effects on health or the environment from disposal of solid waste at such facility. The regulations may provide for the classification of the types of sanitary landfills. Under Sections 4003, 4004, and 4005, open dumps are prohibited in the United States. Existing sites must be either closed or upgraded, and all new sites must be sanitary landfills. Implicit in such criteria is that a reasonable monitoring scheme exist to determine compliance.

The criteria will probably address such impacts as air pollution, surface water contamination, safety, vectors, bird hazards, methane migration hazards, and ground water contamination. A particularly difficult problem area, and subject of this paper, is criteria for ground water. The development of ground water criteria must draw upon the results of research, as being reported at this symposium, and the best thinking of ground water, landfill, attenuation, and leachate experts, many of whom are gathered here today.

Leachate, when generated, must go somewhere. Sometimes it seeps into surface waters where it is usually rapidly diluted and dissipated. In stagnant, slow moving, or low volume waters, however, leachate may have a significant impact. Our leachate damage assessment study identified 47 cases of leachate-related fishkills in 20 states from 1963 to 1974 involving over 65 miles of streams and 42 acres of lakes and killing at least 215,000 fish. Surface seeps of leachate, however, can be rather easily prevented or corrected.

Leachate frequently seeps into ground waters. Depending on the underlying soil and ground water conditions, it may be somewhat attenuated and diluted, or it may flow in a concentrated form. It may seep slowly and build up in a perched zone in the case of a bathtub-shaped clay, or it may rapidly flow from the site in the case of fractured bedrock. Unfortunately there is little ground water monitoring at disposal sites in this country. When a downgradient or nearby drinking water well exists, it frequently becomes the first indicator of ground water contamination. Our leachate damage assessment study identified 36 municipal disposal sites in 21 states which are known to have contaminated drinking water wells, and at least 2 more sites are threatening. Six of these cases were investigated in depth. A total of 52 residential wells, 13 public supply wells, and 7 industrial wells were impacted with a total direct damage cost of $3.2 million at these six sites (excluding the costs of the wells themselves). The major expenditures were investigative and avoidance costs (the contaminated ground water was abandoned or avoided and replacement water was piped in). In none of the cases has the source of contamination been corrected. In one of these cases involving 25 residential wells, the cost for a replacement supply, excluding the cost of inconvenience, etc., was over $20,000 per affected home. In one case over $2 million has been spent already and another $8 million is possible to correct the problem. Another case, not one of the six in-depth studies, is estimated to cost $4.3 million if deep-well injection of contaminated water is permitted, otherwise more than $18 million to correct the problem if leachate treatment is required. The cost to move or retrofit a site is very high. In Buffalo, New York, a landfill containing 2 million cubic yards was moved (not because of a leachate problem) by conveyors and barges to another site 6 miles away at a cost of $10.25 million.

Fifteen sites in eight states and three foreign sites have been identified where the extent of leachate migration was determined. In the United States, leachate from two sites has migrated 2 miles and from another site 1 mile. In Krefeld, Germany, more than 5 miles from one site (over an 18-year period) was reported. Frequently, the extent of migration in ground water is cut off by ground water discharge into surface waters.

The characteristics of leachate are primarily a function of the types of waste, amount of infiltrating water, and pH. High concentrations of heavy metals, other inorganics, organics, and biological contaminants, are common in raw leachate. The constituents that migrate the furthest are those which dissolve readily in water and have a high ion exchange capacity. These parameters are mostly aesthetic impactors on ground water (Cl, Na, K, etc.). The heavy metals (Pb, Hg, etc.), complex organics, and bacteria, many of which have health implications, apparently tend not to migrate very far unless there is a highly permeable soil or free-channel flow as in fractured bedrock.

Certainly this data demonstrates adverse effects on the environment, but how does one write criteria to minimize the probability of their occurrence?

There are two general types of criteria, each with its advantages and disadvantages: (1) performance and (2) operational criteria. Performance criteria specify environmental limits. A continuum of possibilities exists, including zero discharge, drinking water limits, significant increase over background, and specific levels for different classes of facilities or areas based on different geologic, climatic, or water resource conditions. Operational criteria, including "Best Management Practices" or "Maximum Practicable Technology", may specify: ways to minimize infiltration (e.g., control of runon/runoff, cover material, etc.); site location (e.g., out of wetlands and flood plains, near discharge zones, etc.); depth to ground water; thickness of certain soils beneath sites; or liners, etc. Both types already exist in various state regulations.

The selection of the type of criteria must be based on: (1) ability to predict environmental impact or performance of specific operational criteria or technologies, (2) availability (and cost) of effective preventive (control) and corrective (retrofit or treatment) technology, and (3) ability to monitor and determine compliance.

If performance criteria are written, a state permit agency still needs to be able to predict for a new site the effect of certain operational technologies or criteria. If operational criteria are written they must with reasonable probability protect the health and environment, i.e., achieve performance criteria. That is, a cause-effect relationship must be established no matter what the type of criteria.

One approach we are considering is to write performance criteria (under §4004) and operational guidelines (under §1008). Performance criteria is based on environmental impact assessments. The operational guidelines would describe the various technologies and costs to achieve the performance criteria. Operational criteria are more likely to need updating and many equally effective or site specific operational options may exist.

We are also considering the classification of geographical or hydrogeological areas. Class I would be critical areas (e.g., certain wetlands, recharge zones of major aquifers, permafrost areas, etc.) for which zero discharge or degradation would be required. This means either (1) do not put the site there, (2) line the site and collect and treat leachate, (3) prevent leachate generation, or (4) utilize existing natural features and special design (e.g., a counterpumping program). The states, which are the enforcement agencies for the open dump prohibition provisions of RCRA, would be able to add to the list of critical or Class I areas, and to designate major aquifers. The less sensitive Class II and Class III areas would have other limits or may have operational criteria and may utilize natural attenuation and mixing zones to achieve them, assuming a reasonable attenuation/pollutant transport model can be developed.

Compliance monitoring is a problem because of the site-specific inexact nature and cost of monitoring, the generally slow-moving, slow-cleansing characteristics of ground water, and the high cost and uncertainties of corrective technologies. A typical question is "Why monitor; if we find ground water contamination, we do not know what to do about it anyway?"

There are three major purposes for monitoring: (1) as a warning mechanism for ground water users (to show existence of contamination and rate of leachate movement), (2) to measure technological effectiveness (cause and effect), and (3) for evidential/compliance data for possible enforcement and remedial program designs.

When and how to monitor to determine compliance should probably be specified by EPA. However, a cookbook approach is impractical since monitoring systems must be tailored to the conditions at each site. Only general monitoring guidelines are appropriate.

Through our monitoring efforts we have identified a number of monitoring problems that may inhibit compliance investigations. We have also developed some ideas on monitoring. Some basic questions on compliance monitoring for performance criteria include:

(1) Where should ground water impact be monitored? Beneath the wastes, or downgradient? How far downgradient; at the property line? How large an attenuation/mixing zone should be permitted?

(2) Where should upgradient (background) wells be located? How many? What depths? Use cleanest upgradient or average of upgradient for comparison with downgradient wells? What if the site is located on a ground water divide, and therefore has no "true" background?

(3) Where should downgradient wells be located? How many? What depths? Use worst downgradient or average of downgradient; worst well or worst parameter reading?

(4) What if the site is near a discharge zone and it is physically impossible to get a drill rig downgradient?

(5) What about seasonal fluctuations? Use worst level, average, or other statistical method?

(6) What constituents or parameters should be measured as leachate indicators or for determining potential health effects?

Our current approach is to use electrical resistivity survey to identify potential contamination enclaves. If the resistivity works it is useful in locating upgradient and downgradient wells. Available site information and surface observation, such as reported earlier by Dwight Sangrey, are used to estimate ground water flow and stress areas, etc. We do not recommend throughfill wells for compliance monitoring. At minimum, one to three background wells are used and three to five downgradient wells. Indicator parameters (Cl, Fe, COD, Specific Conductance) are monitored periodically. A one-tailed student t-test is used to compare the data from the worst downgradient well (presumably nearest the worst part of the plume) with what appears to be a reasonable background well. If the t-test shows a statistically significant difference then health parameters are monitored, and all known information on the site is used to determine if the disposal site is indeed the most likely cause of the statistical difference.

In summary, all the past and current research on leachate generation, characterization, attenuation, migration, treatment, control, and monitoring must be used as the basis for the criteria. An option we are considering because of the short statutory timetable of one year is to publish interim criteria this year and update this later when the results of current research are in and when the success of the interim criteria is determined. We need your help and guidance in the development and review of the criteria. We plan to send preliminary drafts to our Regional Offices, to all states, and to selected "experts" in the field, but any comments/ advice beforehand are appreciated.

ATTENUATION OF LEACHATE POLLUTANTS BY SOILS

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INTRODUCTION

In 1972 the Solid and Hazardous Waste Research Division of the USEPA Municipal Environmental Research Laboratory in Cincinnati initiated two studies of municipal solid waste leachate pollutant movement in soil. The objectives of these contracts were to determine the important mechanisms controlling municipal pollutant attenuation and movement in soils and to develop methods for predicting pollutant behavior in soils. These studies have been completed and although the final reports have not yet been published, most of the results have been published in the open literature. This paper lists sources of available information on these projects and discusses some of their significant findings. The majority of the information in this paper was drawn from draft final reports submitted by the projects and from the sources cited in the bibliography at the end of this paper.

Because it was assumed that clay would be the fraction of the soil most active in pollutant attenuation the work was split into two projects. The work at the University of Arizona, Tucson, directed by Dr. Wallace Fuller (Contract 68-03-0208), dealt with representative whole soils from the major soil groups in the United States while the work at the Illinois State Geological Survey, Urbana, directed by Dr. Neil Shimp and Dr. Robert Griffin (Contract 68-03-0211), dealt with mixtures of pure clay minerals. The soils used at Arizona were collected from the B-horizon or below to avoid high organic matter contents that would be untypical of soils below landfills. The clays used at Illinois were commercially available materials (e.g. southern bentonite) that were subsequently purified by sedimentation, calcium saturated, and mixed with sand to give materials with a range of clay content and clay mineral type. Some of the physical and chemical characteristics of these materials are summarized in Tables 1 and 2.

The municipal leachates used by Illinois were collected from landfills in the area. Landfill leachate is not available in Arizona because rainfall is so low in comparison to potential evaporation. Consequently, the Arizona project used an above-ground tank and representative municipal refuse to generate the leachate used in their work. The ranges of characteristics for leachates used at Arizona and Illinois are listed in Table 3.

TABLE 3. LEACHATE CHARACTERISTICS

Parameter	Arizona (Dupage)	Illinois (Blackwell)
pH	6.6-6.8	6.8
COD (mg/l)	160-200	1340-1362
Conductivity (m mho/cm)	2.4-2.6	7.2-10.2
Fe (mg/l)	60-120	3.0-4.4
Ca (mg/l)	160-225	46.8-49
K (mg/l)	850-950	491-516

Both projects packed the soils and soil materials into columns, applied leachate anaerobically to the columns with a continuous flow, saturated system, and measured contaminant concentrations in the applied leachate and in the column effluents. After a suitable period, the columns were dismantled and the soils sectioned and analyzed to determine the distribution of contaminants retained by the soils. In the

TABLE 1

Some Characteristics of the Soils Used in Research at the University of Arizona

Series	Order[1]	Soil Paste pH	Cation Exchange Capacity	Electrical Conductivity of Extract	Column Bulk Density	Surface Area	Free iron oxides	Total Mn	Texture[2] Sand	Texture[2] Silt	Texture[2] Clay	Major[3] Clay Minerals
			meq/100g	µhos/cm	g/cm^3	m^2/g	%	ppm	%	%	%	
Wagram	Ultisol	4.2	2	225	1.89	8.0	0.6	50	88	8	4	Kaolinite, Chlorite
Ava	Alfisol	4.5	19	157	1.45	61.5	4.0	360	10	60	31	Vermiculite, Kaolinite
Kalkaska	Spodosol	4.7	10	237	1.53	8.9	1.8	80	91	4	5	Chlorite, Kaolinite
Davidson	Ultisol	6.2	9	169	1.89	51.3	17.0	4100	19	20	61	Kaolinite
Molokai	Oxisol	6.2	14	1262	1.44	67.3	23.0	7400	23	25	52	Kaolinite, Gibbsite
Chalmers	Mollisol	6.6	26	288	1.60	125.6	3.1	330	7	58	35	Montmorillonite, Vermiculite
Nicholson	Alfisol	6.7	37	176	1.53	120.5	5.6	950	3	47	49	Vermiculite
Fanno	Alfisol	7.0	33	392	1.48	122.1	3.7	280	35	19	46	Montmorillonite, mica
Mohave	Aridisol	7.3	10	615	1.78	38.3	1.7	825	52	37	11	Mica, Kaolinite
Mohave (calcareous)	Aridisol	7.8	12	510	1.54	127.5	2.5	770	32	28	40	Mica, Montmorillonite
Anthony	Entisol	7.8	6	328	2.07	19.8	1.8	275	71	14	15	Montmorillonite, Mica

1. U. S. Department of Agriculture Comprehensive Soil Classification System.
2. U. S. Department of Agriculture System: Sand, 2mm - 0.05mm; Silt, 0.05mm - 0.002mm; Clay, <0.002mm diameter.
3. The dominant mineral is listed first.

TABLE 2

COMPOSITION AND SOME PROPERTIES OF THE MATERIALS USED IN THE COLUMN STUDIES AT THE ILLINOIS STATE GEOLOGICAL SURVEY

Treatment[1] %	Treatment[1] Material	Cation Exchange Capacity	Bulk Density	Initial Hydraulic Conductivity[2]
		meq/100g	g/cm^3	cm/sec
100	Sand	0.0	1.71	1.27E-03
2	Montmorillonite	1.7	1.71	9.45E-04
4	Montmorillonite	3.3	1.77	4.34E-04
8	Montmorillonite	6.8	1.79	4.70E-04
16	Montmorillonite	13.3	1.87	1.22E-05
16	Montmorillonite	-	1.93	3.40E-07
32	Montmorillonite	27.3	1.55	1.27E-06
64	Montmorillonite	50.7	1.23	3.05E-07
100	Montmorillonite	79.5	0.84	7.26E-07
2	Kaolinite	0.2	1.68	7.44E-04
4	Kaolinite	0.5	1.76	4.78E-05
8	Kaolinite	1.0	1.80	9.90E-04
16	Kaolinite	2.2	1.87	2.86E-05
16	Kaolinite	-	1.94	1.09E-06
32	Kaolinite	4.3	1.66	2.40E-06
64	Kaolinite	8.2	1.22	5.45E-07
100	Kaolinite	15.1	0.90	2.98E-07
4	Illite	0.7	1.80	8.17E-04
16	Illite	2.7	1.83	2.68E-05
8	Montmorillonite + 8 Kaolinite	7.6	1.95	5.35E-07
8	Kaolinite + 8 Illite	2.8	1.95	1.48E-06
8	Kaolinite + 8 Illite + 8 Montmorillonite	9.2	1.64	8.08E-06

[1] Quartz sand added to make 100%

[2] Exponential notation: E-03 means 10^{-3}

Arizona study high concentrations of single contaminants (As, Be, Cd, Cr, Cu, Pb, Hg, Ni, Se, V, Zn) were added to the leachate before it was applied to the soils; work focused on the movement and retention of these in soils. In the Illinois study the leachates were applied without any additions; they contained very low levels of heavy metal contaminants. Consequently the Illinois column study focused on the movement and retention of the more common leachate contaminants (Na, K, Ca, Mg, Fe, Mn, Cl, B, NH_4, COD) and only limited information was obtained on the heavy metal contaminants Pb, Zn, Cd, and Hg. Additional batch work on pollutant retention was carried out by both projects.

The Arizona project conducted sequential extractions of the soils from the columns to determine not only the vertical distribution of retained contaminants but also the strength with which these contaminants were held by the soil. Each section of soil from the column was extracted first with water and then with 0.1 N HCl to identify the readily leachable materials and those that were held so strongly that they would be nearly unavailable for leaching.

The Illinois project conducted a series of batch sorption tests to study the capacity of the pure clays Montmorillonite and Kaolinite to remove the contaminants Pb, Cu, Zn, Cd, Cr, As, Se, and Hg from aqueous solution and from several landfill leachates. The major variables in these single element studies were pH and the concentration of the contaminant. In addition to identifying the removal capacities of clays for contaminants in a leachate matrix, this work also provided information on the correct procedures for conducting such studies.

The work on both projects has been completed and draft final reports have been submitted. Because some delay is expected in publishing these final reports, a bibliography has been compiled of interim publications by EPA and papers in various journals relating to these two projects. The bibliography is listed at the end of this paper. The first two publications, the symposia proceedings, are available either from:

Technical Information Operations Staff
U.S. Environmental Protection Agency
26 West St. Clair Street
Cincinnati, Ohio 45268

or from the National Technical Information Service. The NTIS numbers and prices for paper (p) or microfiche (mf) are listed in parentheses in the description of the symposia proceedings in the bibliography. The NTIS address is:

National Technical Information Service
U.S. Department of Commerce
Springfield, Virginia 22161

CONCLUSIONS OF THE ILLINOIS PROJECT

The Illinois project concluded that Cl, Na, and water soluble organic compounds (COD) were relatively unattenuated by passage through the columns of clay-sand mixtures; K, NH_4, Mg, Si, and Fe were moderately attenuated. The cation exchange capacity of the clay minerals was concluded to be the dominant removal mechanism for these substances. Heavy metals such as Pb, Cd, Hg, and Zn were present in the leachate at very low levels; these metals were nearly immobile in the columns. The principal attenuation mechanism for the heavy metals was precipitation with resultant accumulation in the surface layers of the columns.

As an aid in organizing and presenting the results, an attenuation number was calculated for each contaminant and treatment. The ratio (C/C_0) of the contaminant concentration measured in the column effluent to the contaminant concentration present in the leachate added to the column was plotted graphically as a function of the number of pore volumes of column effluent. Data for the first pore volume was not included in the calculations because it represents displacement of deionized water present initially in the column. The area above the curve, between one and eleven pore volumes, expressed as a percentage of the total area bounded by one to eleven pore volumes and a C/C_0 of zero to one, was taken as the attenuation number. In this system, contaminants which were strongly retained by the soils had high attenuation numbers while those which passed readily through the columns had low numbers. Greater amounts of B, Mn, and Ca were eluted from the columns than were applied in the leachate. Attenuation numbers for these constituents were reduced by 100 and a negative sign was prefixed to indicate net release.

The low mobility contaminants Pb, Zn, Cd, and Hg had mean attenuation numbers ranging from 96.8 to 99.8. The moderate mobility contaminants Fe, K, NH_4, and Mg had numbers ranging from 29.3 to 58.4. The high mobility materials COD, Na, and Cl had numbers ranging from 10.7 to 21.3. Boron, Mn, and Ca were eluted from the columns in amounts greater than present in the leachate and all had negative numbers.

Aluminum, Cu, Ni, Cr, As, SO_4, and PO_4 were present in the DuPage leachate in such low concentrations that they were attenuated nearly completely and never appeared in the column effluents in significant concentration. Although these constituents are expected to have different mobilities, this cannot be demonstrated with the data from the column effluents.

In the batch studies the adsorption in leachate was 50 to 90% lower in most cases than the clays' adsorption capacity for the metal ions in pure aqueous solutions. The pH of the leachate significantly affected the amount of attenuation. It was concluded that the heavy metal cations Pb, Cd, Cu, Cr(III), Hg, and Zn attenuated primarily by an exchange-adsorpotion mechanism which was affected by pH and competition from other cations. As the pH increased to values of 5-6, a large increase in removal was observed due to increased adsorption of metal complex ions and formation of insoluble heavy metal hydroxide and carbonate compounds. At high pH the primary mechanism of attenuation for these ions was precipitation.

To illustrate the magnitude of the matrix-competition effect on sorption some data for sorption of Pb as $Pb(NO_3)_2$ on kaolinite is listed below.

Solution Matrix	Sorption (ug/g)	% of Maximum
Deionized Water	15,800	100
0.1 M NaCl	11,100	70
DuPage Leachate	5,000	32
Blackwell Leachate	1,500	10

These differences in sorption from different matrices are likely due to interference in the exchange process by other solutes in the matrix competing for exchange sites on the clay. The 0.1 M NaCl solution contains only one competing cation while the leachates, particularly the Blackwell contain many cations in high concentrations which could compete with the added Pb for sorption sites. The DuPage is an older (weak) leachate that was used in the column studies and batch adsorption studies. Its properties are listed in Table 3. The Blackwell is a very strong (young) leachate that was used in only a few of the batch studies.

The effect of pH on the attenuation of the heavy metal anions Cr(VI), As, and Se was found to be the opposite of the cations and it was concluded that precipitation was not an important mechanism for the anions. Because the adsorption of the anions correlated well with the distribution in solution of the ionic species $HCrO_4^-$, $H_2AsO_4^-$, and $HSeO_3^-$ it was concluded that the principal attenuation mechanism for the heavy metal anions was adsorption of the monovalent species. Because of the relative magnitude of adsorption and their pH- related behavior, the heavy metal anions would be significantly more mobile at higher pH values than the cations.

Measurements of the formation of Pb, Cd, and Hg organic complexes in leachates suggested that complexation was of secondary importance to adsorption and precipitation. This is likely due to competition from high concentrations of other cations present in leachates.

Significant reduction in hydraulic conductivity of the materials in the column studies was concluded to be due to microbial activity. These observations suggest that the hydraulic activity of clay-sand liners placed in the bottom of a landfill will decrease with continued exposure to municipal leachate.

CONCLUSIONS OF THE ARIZONA PROJECT

The major conclusions from the work at Arizona are as follows:

1. Soil properties most useful in predicting attenuation (retardation of migration) of contaminants by soils are:
   a. Clay content,
   b. content of hydrous oxides, primarily iron oxides,
   c. pH and content of free lime, and,
   d. surface area per unit weight of soil.

2. The mobility of the eleven contaminants studied may be classified as follows:
    a. most generally mobile - Cr(VI), Hg, Ni
    b. least generally mobile - Pb, Cu
    c. mobility varies with conditions - As, Be, Cd, Se, V, Zn.

3. The characteristics of the leachate, primarily electrical conductivity and total organic carbon (TOC), significantly affect the migration of contaminants.

4. Soils have a variable but significant capacity to retain sorbed contaminants against extraction and further leaching.

Some of the data supporting these conclusions is summarized in Tables 4, 5, and 6. Table 4 lists the correlation between soil properties and the mass of contaminant removed per gram of soil in the column, per ml of leachate applied to the column. This is a measure of the soil's ability to at least temporarily attenuate the trace element. Data for Pb, Cu, and Hg are not shown; Pb and Cu were so immobile that they did not migrate appreciably while Hg migrated so rapidly that differences between soils were slight. The clay content of a soil and the surface area per unit weight were by far the best single predictors of a soil's attenuation properties. The cation exchange capacity and content of free iron oxides were also useful, though much less than clay and surface area. Correlations for surface area were best for those elements added to the leachate as divalent cations while the correlations for free iron oxides were best for elements (except for arsenic) added as anions. Based on information in the literature, the preferential sorption of As by hydrous oxides was expected. The lack of a significant correlation between As removal and content of iron oxides may be due to interferences of the leachate organic fraction in the sorption process.

The usefulness of combinations of soil properties as predictors of attenuation was analyzed via stepwise linear regression. A regression equation using clay content, surface area, and iron oxide content as independent variables will predict removal of Cd, Be, Ni, and Zn with significance at the 1% level. The correlation coefficients for equations relating clay, surface area,

TABLE 4

CORRELATION[1] BETWEEN SELECTED SOIL PROPERTIES AND

MASS OF CONTAMINANT RETAINED[2]

ELEMENT	SOIL PROPERTIES				
	Clay Content	pH	Cation Exchange Capacity	Surface Area Per Gram	Free Iron Oxides
Arsenic	0.88^{++}	0.22	0.42	0.66*	0.60
Beryllium	0.78**	0.47	0.62*	0.81**	0.52
Cadium	0.67*	0.48	0.60	0.71**	0.46
Chromium	0.56	-0.43	0.21	0.10	0.75**
Nickel	0.59*	0.51	0.79**	0.88^{++}	0.27
Selenium	0.71*	-0.30	0.44	0.39	0.68*
Vanadium	0.84^{++}	0.16	0.55	0.61*	0.59*
Zinc	0.83**	0.52	0.71*	0.84**	0.50

1. Degrees of freedom are not the same for all correlations.
2. Per gram of soil in the column, per ml of leachate applied to the column.
* Significant at the 5% level
** Significant at the 1% level
++ Significant at the 0.1% level

TABLE 5
THE AMOUNT OF SORBED TRACE ELEMENT EXTRACTED FROM DIFFERENT SOILS BY WATER

Soils	Trace Element								
	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn
	%								
Wagram l.s.	0	9	0	.5	1.6	7	0	9	8
Ava si.c.l.	3	5	2	.3	0	6	.7	0	5
Kalkaska s.	2.5	2	.5	.5	0	18	.3	7.5	5
Davidson c.	.5	1	4	1.5	2.2	1	0	.4	3
Molokai c.	0	.5	1	.1	0	1.1	0	.7	2
Chalmers si.c.l.	-	-	-	-	.5	3	-	3.8	-
Nicholson si.c.	2	1	10	0	0	.2	.2	1.5	1
Fanno c.	4.5	1	1	3	.5	2	.2	8	1.2
Mohave s.l.	0	2	0	.5	0	4	.4	.3	2.5
Mohave$_{Ca}$ c.l.	11.5	.2	6	.2	0	.1	.1	23	.1
Anthony s.l.	0	3.5	0	3	9	18	0	28	2.8

TABLE 6
THE AMOUNT OF SORBED TRACE ELEMENT EXTRACTED FROM DIFFERENT SOILS BY 0.1 N HCl

Soils	Trace Element								
	As	Cd	Cr	Cu	Hg	Ni	Pb	V	Zn
	%								
Wagram l.s.	11	90	8	60	14	100	41	100	100
Ava s.c.l.	11	85	7	40	16	72	78	75	64
Kalkaska s.	18	100	16	67	31	100	70	48	100
Davidson c.	5	77	7	67	49	43	70	36	42
Molokai c.	0	87	5	57	59	78	59	35	100
Chalmers si.c.l.	-	-	-	-	22	57	-	80	-
Nicholson si.c.	11	75	13	27	24	63	47	55	64
Fanno c.	24	80	8	44	17	58	58	89	66
Mohave s.l.	11	89	7	55	30	80	77	93	100
Mohave$_{Ca}$ c.l.	20	74	17	51	59	31	57	77	41
Anthony s.l.	13	100	100	68	23	100	93	63	100

and iron oxide with V and As are lower but are still significant at the 5% level. Inclusion of soil pH as an independent variable improved the significance of the correlation only for Cr and Se, elements that are present in solution as anions.

Data on the amounts of trace elements extracted from the soils after completion of the column studies is listed in Tables 5 and 6. The acid extraction removed much greater amounts than the water extraction. The substantial amounts of sorbed trace elements not removed by even the acid extraction suggest a significant permanent attenuation capacity for many soils.

As a qualitative summary, Figures 1 and 2 present a ranking of the soils used in the project according to their attenuation properties and a ranking of the contaminants according to their mobility in the various soils. The physical and chemical properties of these soils are summarized in Table 1. Separate rankings are given for contaminants present in the leachate as cations and as anions because of the differences in migration behavior. Note that every change in the ordering of soils when going from cations to anions involves a higher ranking for soils having a lower pH and/or a higher content of free iron oxides.

It is concluded that clay content, surface area, and content of hydrous oxides and free lime will be the soil properties most useful in selecting safe disposal sites for municipal and hazardous wastes. Additionally, the data suggests that use of lime and iron oxides should be examined as practical management tools for minimizing the movement of contaminants from landfills.

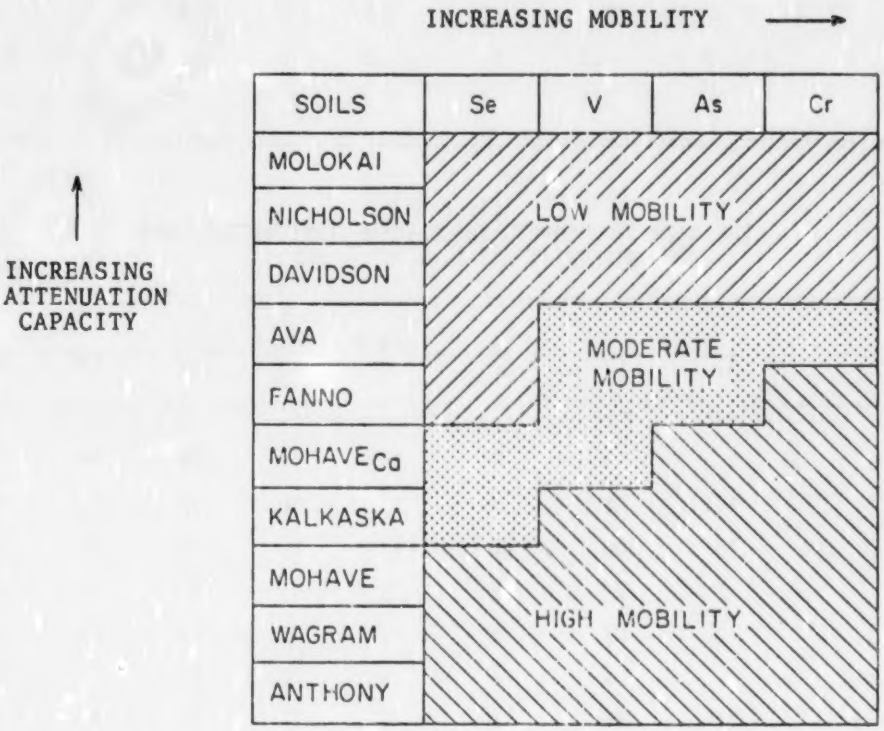
INCREASING MOBILITY →
INCREASING ATTENUATION CAPACITY ↑
SOILS
Se
V
As
Cr
MOLOKAI
NICHOLSON
DAVIDSON
AVA
FANNO
MOHAVE Ca
KALKASKA
MOHAVE
WAGRAM
ANTHONY
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

FIGURE 1

RELATIVE MOBILITY OF ANIONS USED IN THE UNIVERSITY OF ARIZONA STUDY

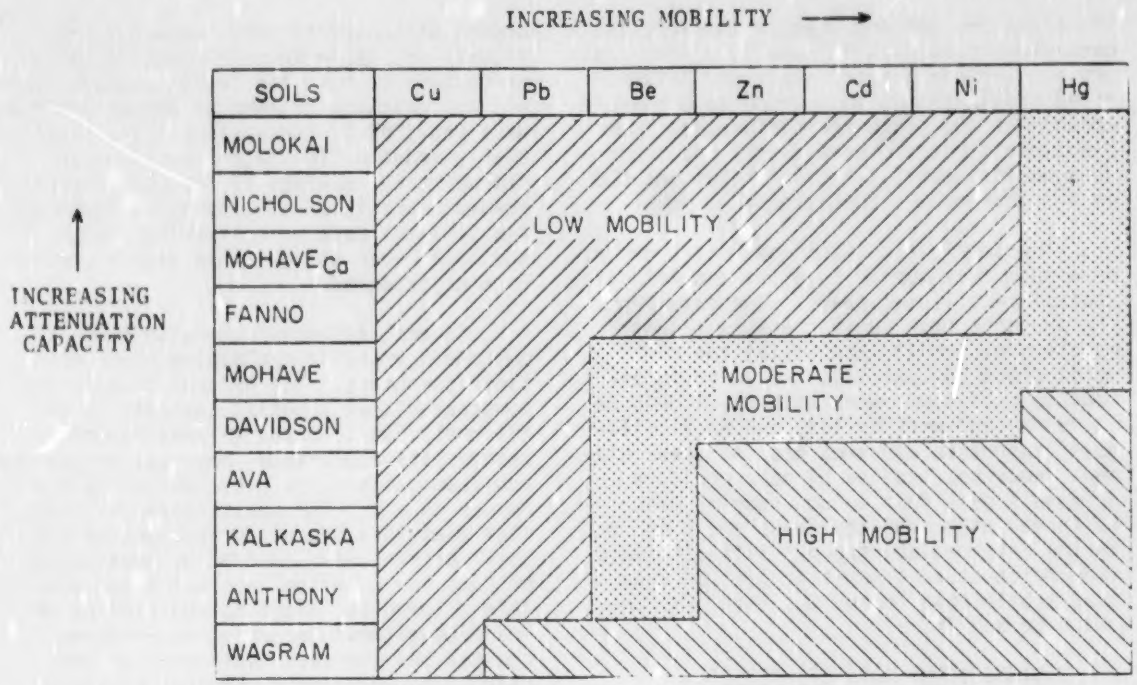
INCREASING MOBILITY
SOILS
Cu
Pb
Be
Zn
Cd
Ni
Hg
MOLOKAI
NICHOLSON
MOHAVE Ca
FANNO
MOHAVE
DAVIDSON
AVA
KALKASKA
ANTHONY
WAGRAM
INCREASING ATTENUATION CAPACITY
LOW MOBILITY
MODERATE MOBILITY
HIGH MOBILITY

FIGURE 2

RELATIVE MOBILITY OF CATIONS IN SOILS USED IN THE UNIVERSITY OF ARIZONA STUDY

COMPARISON OF RESULTS FROM THE PROJECTS

Although the projects used quite different materials and procedures the results of the work are complementary. Both concluded that the clay content and pH of the soil would have a major influence on contaminant retention and movement in soils. Likewise both concluded that leachate characteristics would also have such a major influence that it would be nearly impossible to extrapolate measurements in one leachate-soil system to predict behavior of contaminants in a different leachate-soil system.

Illinois found that below pH 5 exchange-adsorption was the dominant removal mechanism for the metal cations and that as the pH increased through the range of 5 to 6, precipitation became a more significant removal mechanism.

Arizona found weak correlations between cation exchange capacity and contaminant removal by all soils and concluded that CEC would not have significant value for predicting contaminant removal by soils. This was likely due to the masking effect of non-CEC removal processes active in whole soils but not in a "clean" clay-sand system and also due to the average pH of the Arizona soils. The soils used in the Arizona project were neutral to slightly acid and only a few had pH of less than 5. It is possible that a selection of soils with a lower average pH would have given a stronger correlation between CEC and metallic contaminant removal.

Illinois found the following order of mobility for the non-metallic contaminants and iron:

$Cl > Na > COD > Mg > NH_4 > K > Si > Fe$

Since Arizona concentrated on the metallic contaminants no data is available from that project for comparison. However, Arizona noted that although soils removed different amounts of COD, both the differences and the amounts removed were slight and their data would support Illinois' rating of COD as a highly mobile contaminant.

The metallic contaminants, as a group, were much less mobile than the non-metallic contaminants listed above. The data from the projects show Cu and Pb as the least mobile, Hg, Cr(VI), and Se(IV) the most mobile, and Cd, As(III), and Zn of intermediate mobility. Work with Ni, V, and Be (Arizona) and with Cr(III) and As(V) (Illinois) cannot be compared because these elements were not studied by both projects. Arizona's data from column studies places Ni in the high mobility group and V and Be in the intermediate group. Illinois' data from batch studies shows Cr(III) as even less mobile than Pb and Cu; As(V) was between Cd and Zn.

SIGNIFICANT ADVANCES RESULTING FROM THE PROJECTS

Some of the most significant advances resulting from the projects were documentation of previously known effects. For instance, clay content and pH were known to affect the movement of contaminants and knowledge of salt effects and the lyotropic series suggested that sorption of a contaminant would be affected by the characteristics of the leachate in which it was contained. While the projects did not provide fundamentally new information on these subjects they did provide previously unavailable quantitation of the magnitude of the effects.

The contaminant removal data from the Arizona project demonstrates the range of effects, in a whole-soil system, of variations in pH, clay content, and iron oxide content. The general trends of these effects were known but information on their magnitude for specific contaminants was previously unavailable. Because of the range in characteristics of soils included in the study, the data also provides a measure for estimating the contaminant removal capacity of other soils not included in the study. Similarly, the data from the Illinois project on the effect of contaminant concentration, pH, and composition of the carrier solution, though in accord with expectations, were not previously available in sufficient quantity or quality to make the design calculations and explanations of removal mechanisms that were provided by the project. The microbially induced flow rate reductions and the relative rankings of the contaminant removal capacities of clay types shown by the Illinois data were likewise useful additions even though their general trends were as expected.

Several advances were also made in methodology and in explaining observed field phenomena. The Arizona project demonstrated that landfill leachate is an extremely labile material that rapidly changes its appearance, physical properties, and chemical species distribution upon exposure to air. The substantial difficulties that this causes during routine chemical analyses were averted by developing methods for handling leachate under CO_2. This allowed the sampling and storage of leachate without incurring air-induced changes and improved confidence in the results of subsequent chemical analyses.

While conducting the batch sorption studies of contaminant removal by clays, the Illinois project demonstrated that the results of sorption studies are dependent on the weight of clay and the solution volume and concentration, particularly when de-sorbing ions can compete with contaminants for sorption sites on the clay. It was shown that results of previous sorption work were not as expected because the system conditions did not comply with the assumptions of the competitive Langmuir-type equation used to display the data. Indications are that measured contaminant removal capacities of soil materials will be increased when the conditions of the experiment comply with the competitive Langmuir-type assumptions.

The Illinois project observed large amounts of calcium in the early increments of effluent from the column studies. The amount of calcium eluted corresponded very closely to the amount of contaminants removed from the leachate applied to the columns. This suggested that the calcium was released from the clays by exchange with the contaminants and explained the origin of a similar phenomena, the "Hardness Halo" observed (but not explained) in previously reported studies of actual landfills. The Arizona project observed

COD values of 1,000 to 10,000 mg/l in the early effluents from soil columns receiving leachates with COD values of only 160 to 200 mg/l. The "COD Halo", observed in soils ranging in texture from sand to clay, has not yet been reported in field studies. These "halo effects" might serve as early warnings in field monitoring, indicating that the contaminant front is approaching and that subsequent samples should be analyzed in more detail.

RESEARCH NEEDED

As a conclusion to this discussion of the Arizona and Illinois projects, several of the research needs suggested by the project results will be described. COD was only slightly attenuated by any of the soils or soil materials. This suggests that the leachate organic fraction will be a significant contamination problem and that further work should be focused on the movement of total organics and of specific groups of organic materials in soils and soil materials. Closely related to this is the need to develop treatments to be applied at the refuse-soil interface in a landfill to enhance degradation of organics and removal of inorganic contaminants before the leachate passes into the soil below the landfill. One specific treatment, liners of clay-sand or native, high-clay soils, is presently in use for controlling liquid movement from landfills. The project results indicate that, although this treatment would have only a slight effect on leachate organic movement, it would be highly effective for removing inorganic contaminants and should be tested at a larger scale under field conditions. Finally, further work, taking account of leachate characteristics as well as soil characteristics, is needed to develop methods for predicting contaminant movement and retention in soils as a basis for more rational siting and operation of landfills.

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EFFECT OF MUNICIPAL LANDFILL LEACHATE ON THE RELEASE OF TOXIC METALS FROM INDUSTRIAL WASTES

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ABSTRACT

The potential increase in hazard resulting from the co-disposal of industrial wastes with municipal refuse was tested using wastes from several different industries; namely, electroplating waste, inorganic pigment waste, and nickel-cadmium battery production waste. Known weights of each waste were mixed with municipal landfill leachate and water. The samples were extracted for 24 and 72 hours, filtered, and the filtrates analyzed for cadmium, chromium, copper, and nickel by atomic absorption spectrophotometry. The wastes were recovered, mixed with fresh aliquots of municipal landfill leachate or water and re-extracted. This serial batch extraction was carried out seven times. The concentrations of cadmium, copper, and nickel in the municipal landfill leachate extracts were much higher than was found in water extracts. Depending on the waste, metal, and cumulative extraction volume, the increase in solubilization of the metals by the municipal landfill leachate ranged from approximately 100 to 3000 times higher than with water. Chromium was the only exception. The concentration of Cr metal in both solvent extracts was approximately the same (or slightly greater in the water extracts). These findings dramatically demonstrate the potential hazard that may result from the disposing of certain industrial wastes together with municipal refuse. This raises the serious question as to the advisability of co-disposal in general.

INTRODUCTION

As municipal refuse decomposes in landfills, resultant decomposition products can dissolve in the water released from the refuse itself. The volume of this liquid, or leachate, is increased by rainfall, snow melt or runoff, especially in improperly constructed landfills. If the volume exceeds the "field capacity" of the refuse layer, the leachate passes through the refuse into the underlying soil and may eventually find its way into the ground water. If this underground water is used by a municipality for culinary purposes, the entrance of the landfill leachate potentially presents a serious health hazard. In addition, reclamation of the disposal site and water supply may require years and be extremely costly (1).

Industrial waste are often co-disposed with municipal refuse. The Municipal Environmental Research Laboratory, U.S. Environmental Protection Agency, Cincinnati, was concerned that municipal landfill leachate may solubilize toxic metals contained in industrial wastes. This was investigated here by leaching wastes from several different industries; namely, electroplating, inorganic pigment, and nickel-cadmium battery production wastes, using water or municipal landfill leachate. The concentration of cadmium, chromium, copper, and nickel, was measured in the resultant samples.

MATERIEL AND METHODS

Electroplating Waste

The electroplating waste is produced in plating, phosphatizing, and metal cleaning operations. The metals are precipitated from the waste water by the addition of lime or caustic soda. The weights of the four metals in the waste were: cadmium 0.8 percent (w/w), chromium 10.5 percent, copper 3.2 percent, and nickel 1.0 percent.

Inorganic Pigment Waste

Waste water from the manufacturing of inorganic pigment contains both dissolved and suspended solids. After treatment to destroy cyanide and to reduce hexavalent chromium, the pH is raised to eight with lime. A flocculating polymer is added to promote precipitation. The weight of the four metals in the waste were: cadmium 0.17 percent, chromium 7.0 percent, and copper 0.42 percent. (The nickel content was not determined)

Nickel-Cadmium Battery Productive Waste

This waste arises from the washing of electrodes upon which cadmium and nickel had been deposited. The washings are made alkaline (pH 11-12) with sodium hydroxide. This precipitates both metals as hydroxide salts. Most of the precipitates are recovered from the alkaline waste water. However, "fines" are lost as the waste water is pumped to the holding lagoon. The cadmium weight was 51 percent and the nickel 10.1 percent.

Municipal Landfill Leachate

The municipal landfill leachate used in this study was collected from the experimental test cell in Boone County, Kentucky. The leachate is composed of the liquid content of the refuse combined with natural precipation. The leachate has been used in other studies and extensively characterized (2,3) so only pH, specific conductance, acidity/alkalinity, and toxic metal content were measured. A few visual observations were made as to color, turbidity, effect of air exposure and its interaction with the industrial wastes.

The municipal landfill leachate was stored in a carbon dioxide environment at 4°C. When received, the leachate was a light bluish black in color and contained some greyish black suspended solids. The initial pH was 5.6 and the specific conductance 12,650 micromhos. The alkalinity was measured by titrating an aliquot of leachate with 0.102 N HCl. The alkalinity was calculated to pH 4.7 as recommended in Standard Methods(4). The alkalinity was 1.0×10^{-4} equivalent of acid per milliliter of leachate. The only toxic metal found in a significant concentration was zinc (60mg/l). The other metals of interest were very low.

Upon standing, even at 4°C and in a carbon dioxide invironment, the leachate changed. After one week the pH was 5.8. The suspended solids increased and the specific conductance increased to 14,900 micromhos. After several hours exposure to air the color turned dark brown. The suspended solids increased to the point where the leachate was very turbid. Gas was evolved during this time.

These observations and measurements indicate the problems to be encountered while working with municipal landfill leachate. Leachate samples collected at different landfills will vary widely in composition as will leachate collected from the same site at different times of the year.

Extraction of the Waste Samples

The serial batch extraction procedure used in this study is outlined below. A 20 gram portion of waste was weighed into an Enlenmyer flask. Two hundred milliliters water or municipal landfill leachate was added to the flask. The flask was stoppered and mixed for 72 hours. The mixture was filtered with vacuum through a Buchner funnel. The filtrate was filtered a second time through 0.45 micron millipore filter and then analyzed. The waste was recovered from the Buchner funnel filter pad and a second 200 milliliter aliquot of water or municipal landfill leachate added and mixed for 72 hours. The waste was again filtered as described above. This serial batch extraction was repeated seven times.

Analysis

The pH and specific conductance was measured in each extraction sample. The samples were then acidified with HNO_3 to an acidity of one percent. The concentration of cadmium, chromium, copper, and nickel in the waste extracts was measured by atomic absorption spectrophotometry using an air-acetylene flame. The lower detection limits for the metals were 0.02 mg/l for cadmium, copper, and nickel and 0.05 mg/l for chromium. Very little interference in the analysis was found.

RESULTS AND DISCUSSION

Electroplating Waste

Figure 1 is a plot of the serial batch extraction of cadmium from electroplating waste by water or municipal landfill leachate. The extraction number (or batch) is plotted on the X-axis. The serial batch extraction was carried out seven times. The volume is expressed as milliliters per gram waste, e.g.,20 grams waste extracted with 200 milliliters solvent are equivalent to 10 milliliters per gram. The 20 grams of waste were extracted by a total of 1400 milliliters solvent (70 milliliters per grams of waste). The batch serial extraction can also be expressed in terms of equivalent months. This is intended to show the time that would have elapsed if the waste were continuously leached by one of the above solvents at a rate of 1.3×10^{-4} centimeters per second.

The concentration of cadmium found in each batch sample is plotted on the y-axis (note the y-axis is logarithmic). The dashed line histogram represents the concentration of cadmium found in the samples extracted by water. The solid line histogram is the concentration of cadmium found in the municipal landfill leachate extracts of the waste.

The results show the municipal landfill leachate was far more efficient than water in extracting cadmium from the electroplating waste. The total weight of cadmium extracted from each gram of waste by water was 69 micrograms. Municipal landfill leachate extracted about 70 times more or 4,380 microgram cadmium per gram waste.

Figure 2 is a plot of chromium extracted from electroplating waste by both solvents. As compared to cadmium, a very small amount of chromium was extracted. However, municipal landfill leachate extracted more than twice as much chromium as water. Although a substantial quantity of chromium was present in the waste, most was probably chromium hydroxide. This form of chromium is very insoluble.

Figures 3 and 4 are plots of the weight of copper and nickel extracted from the electroplating waste by municipal landfill leachate and water. The municipal landfill extracted over 1,000 times more copper (7,961 micrograms per gram waste) than water (7 micrograms per gram). It also extracted more than 24 times more nickel than water (965 micrograms per gram compared to 39 micrograms per gram).

The electroplating waste was fairly soluble in water. Approximately three percent was dissolved in the first batch. However, the soluble ions in the subsequent extracts dropped off rapidly, decreasing from specific conductance of 6,400 micromhos in the first extract to 2,200 micromhos in the seventh extract. In contrast, the specific conductance of the municipal landfill leachate extracts remains very high throughout all the serial extractions.

Inorganic Pigment Waste

Figures 5 through 8 are plots of cadmium, chromium, copper, and nickel extracted from inorganic pigment waste by municipal landfill leachate and water. Over 300 times more cadmium was extracted by municipal landfill leachate than water (626 micrograms per grams compared to 2 micrograms per grams). Copper was not detected in the water extracts of the waste. However, a total of 25 micrograms copper per gram waste was found in the municipal landfill leachate extracts. In addition, at least 47 times more nickel was extracted by the municipal landfill leachate. Only chromium was extracted by water in larger amounts (approximately four times) than by municipal landfill leachate.

The inorganic pigment waste was much less soluble in water than was electroplating waste. Approximately one percent of the waste dissolved in the first extraction step and dropped rapidly in subsequent extractions. However, the weight of material extracted by municipal landfill leachate was high even after seven extractions.

Nickel-Cadmium Battery Production Waste

Figure 9 is a plot of the cadmium extracted from the nickel-cadmium battery production waste by water and municipal landfill leachate. Water extracted only a total of 131 micrograms per gram of waste. However, landfill leachate extracted nearly 870 times more for a total of 114,490 micrograms. The extraction of nickel from this waste was much smaller than cadmium as shown in Figure 10. However, the municipal landfill leachate extracted at least 200 times more nickel than water.

Nickel-cadmium battery waste was the most soluble waste used in this study.

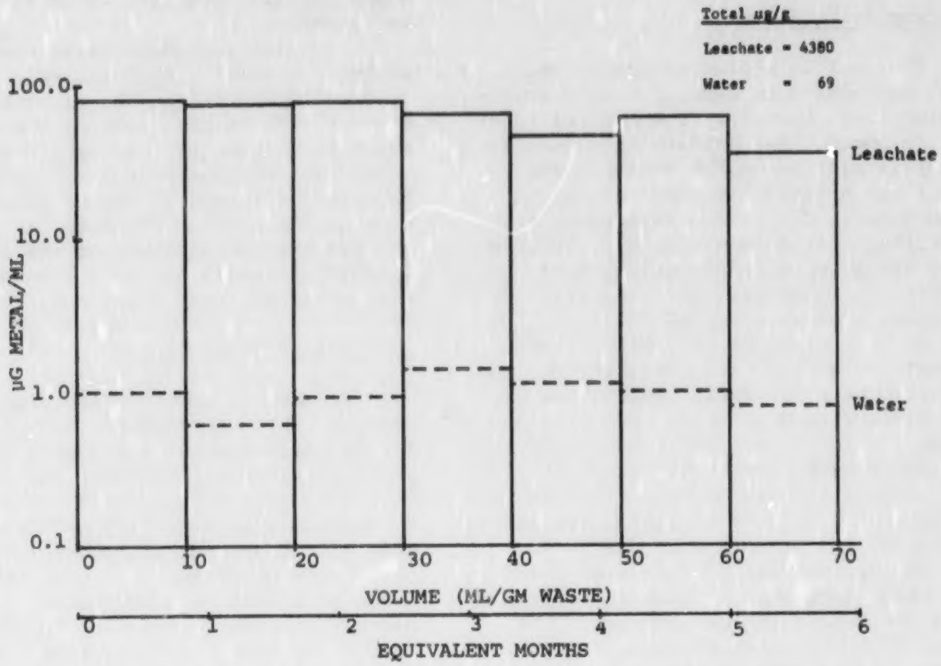

Figure 1. Comparison of Solubilization of Cadmium from Electroplating Waste by Landfill Leachate and Water

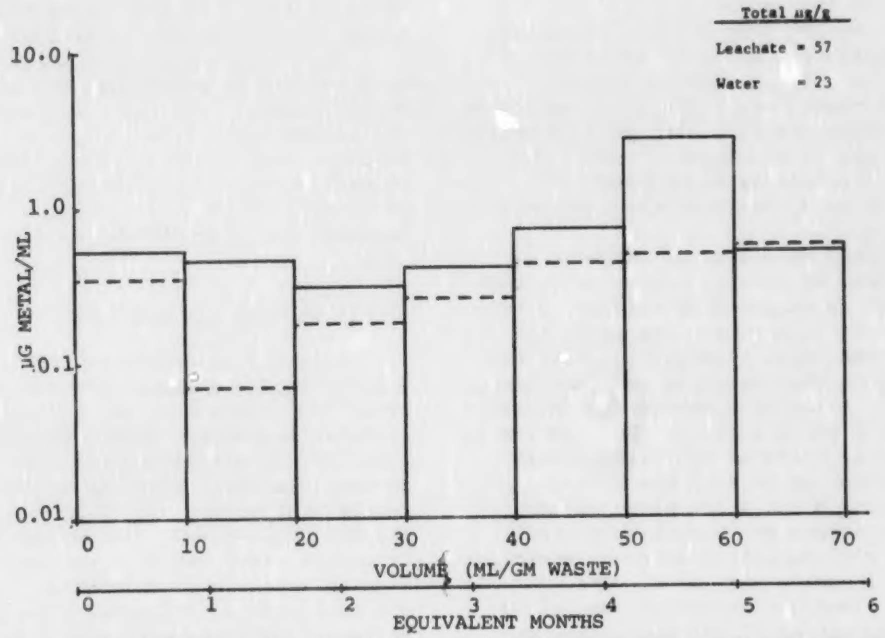

Figure 2. Comparison of Solubilization of Chromium from Electroplating Waste by Landfill Leachate and Water

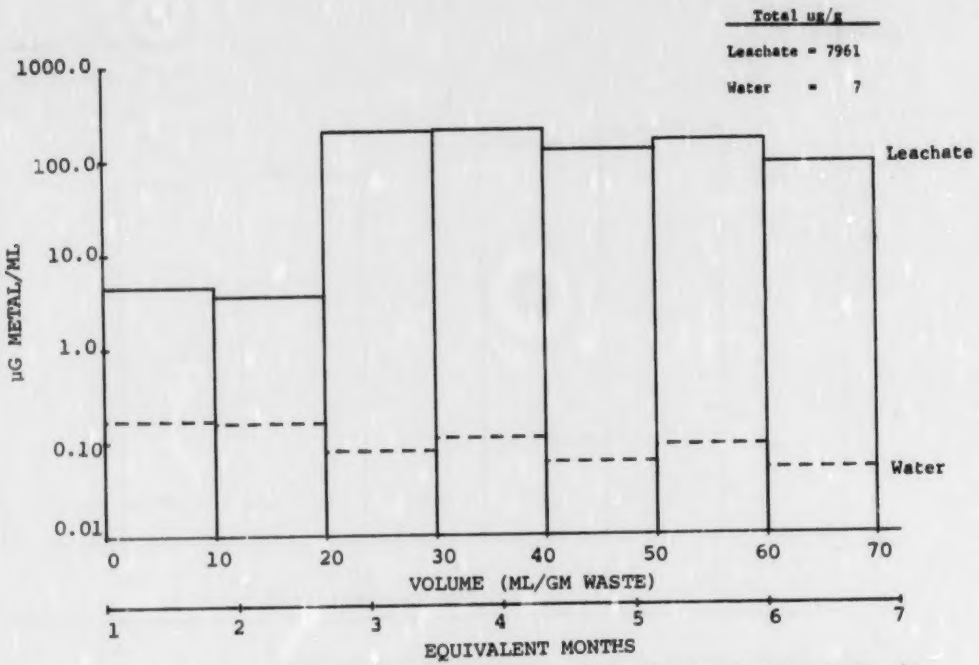

Figure 3. Comparison of Solubilization of Copper from Electroplating Waste by Landfill Leachate and Water

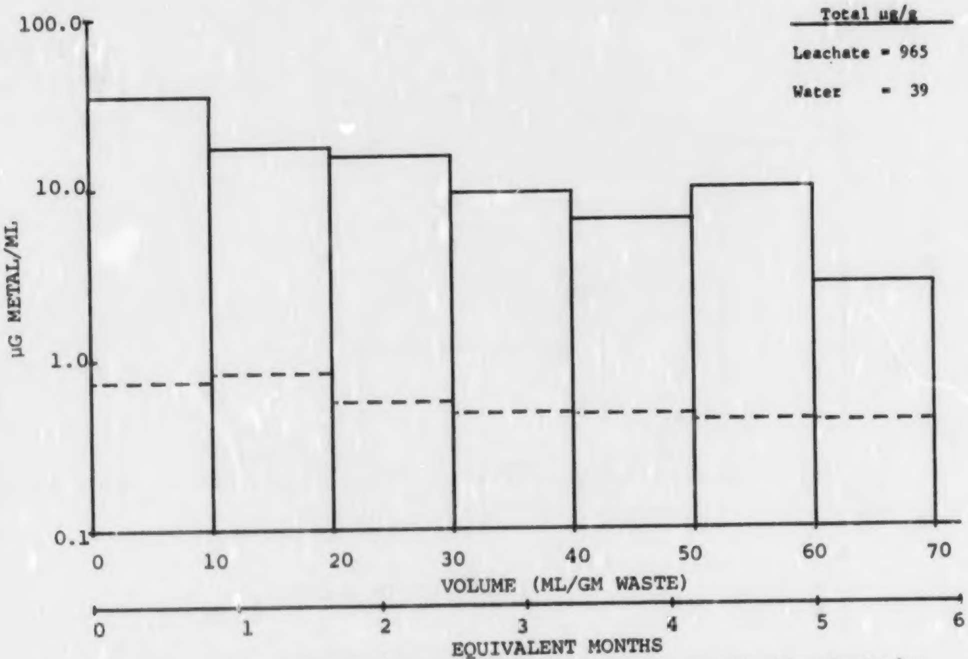

Figure 4. Comparison of Solubilization of Nickel from Electroplating Waste by Landfill Leachate and Water

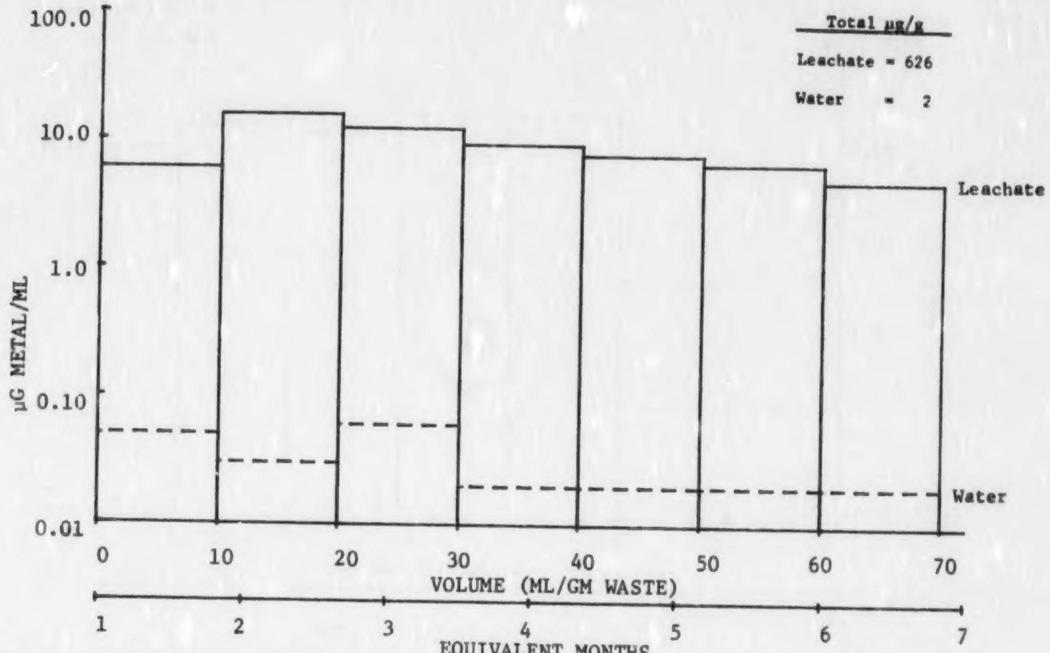

Figure 5. Comparison of Solubilization of Cadmium from Pigment Waste by Landfill Leachate and Water

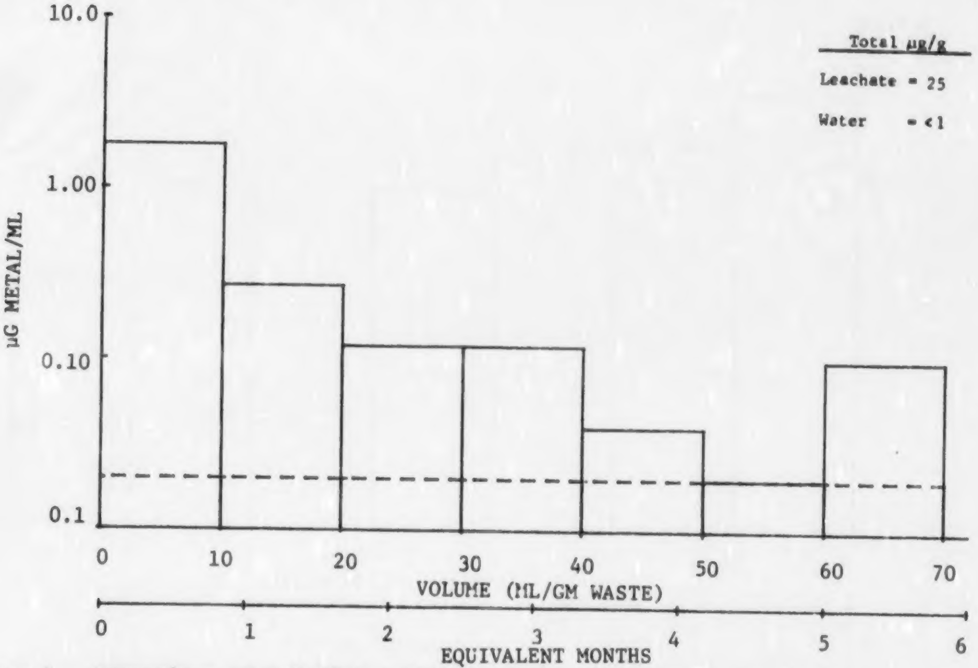

Figure 6. Comparison of Solubilization of Copper from Pigment Waste by Landfill Leachate and Water

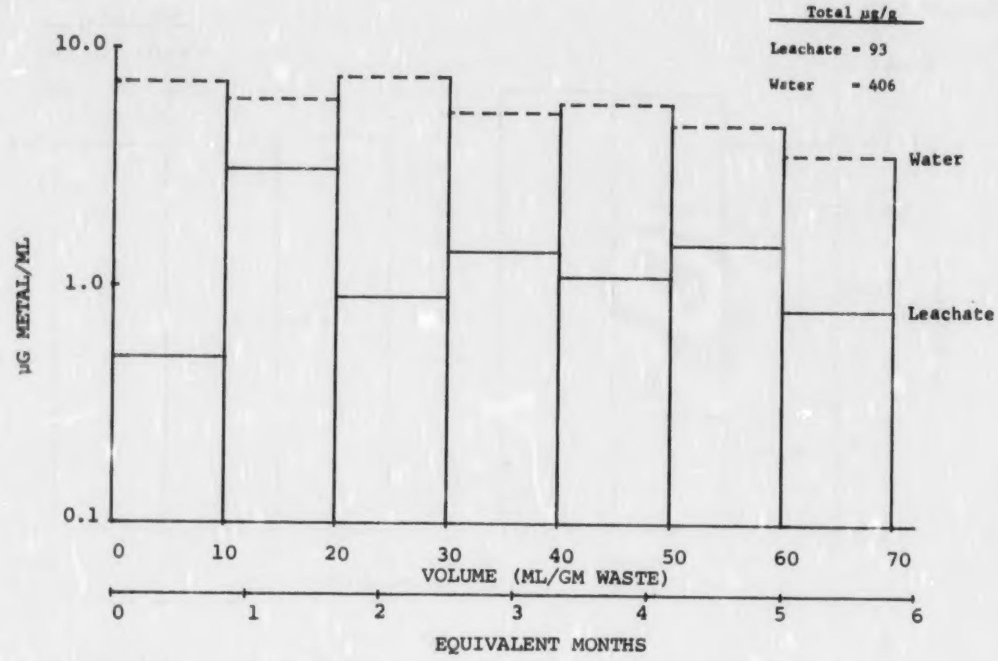

Figure 7. Comparison of Solubilization of Chromium from Pigment Waste by Landfill Leachate and Water

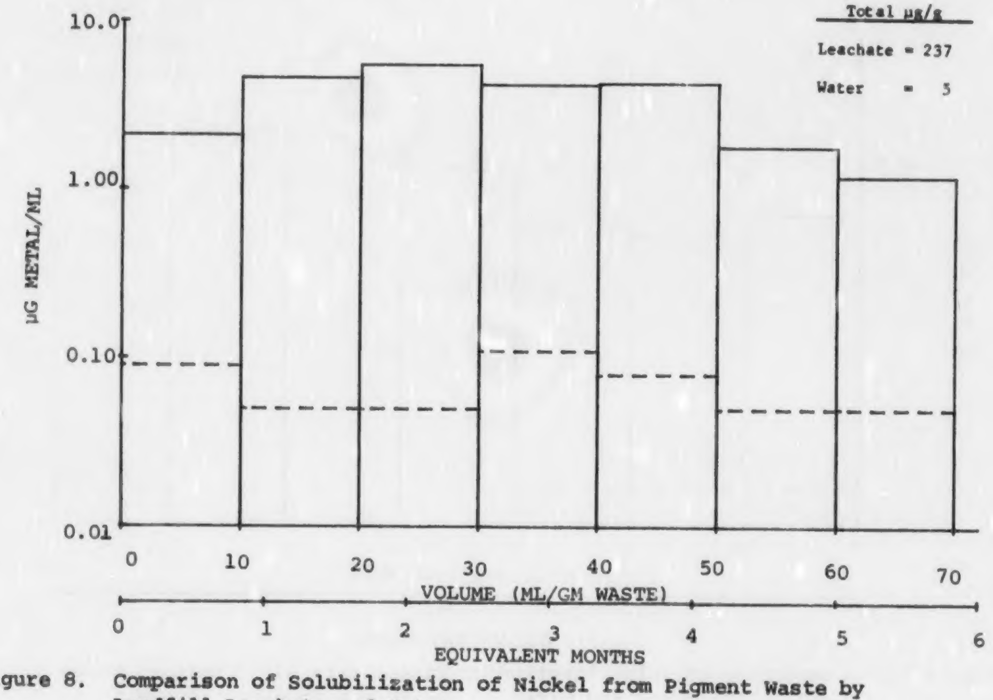

Figure 8. Comparison of Solubilization of Nickel from Pigment Waste by Landfill Leachate and Water

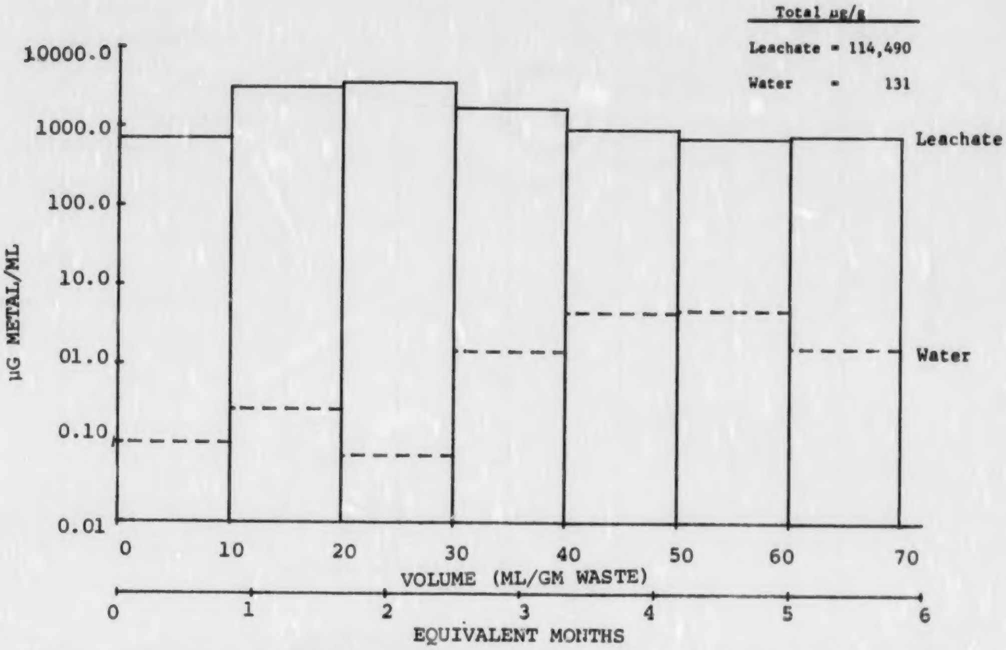

Figure 9. Comparison of Solubilization of Cadmium from Ni-Cd Battery Waste by Landfill Leachate and Water

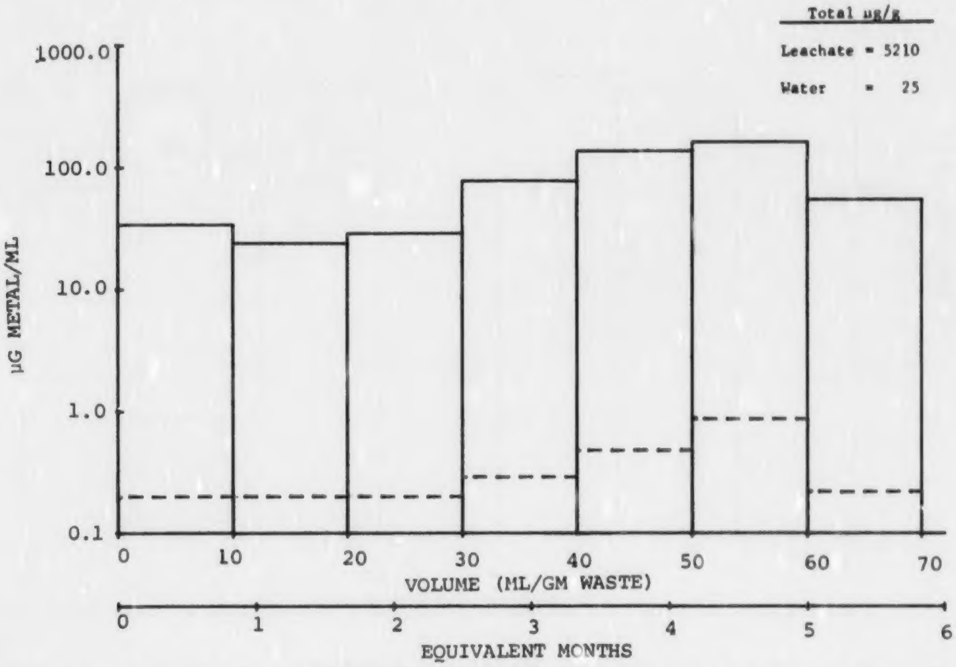

Figure 10. Comparison of Solubilization of Nickel from Ni-Cd Battery Waste by Landfill Leachate and Water

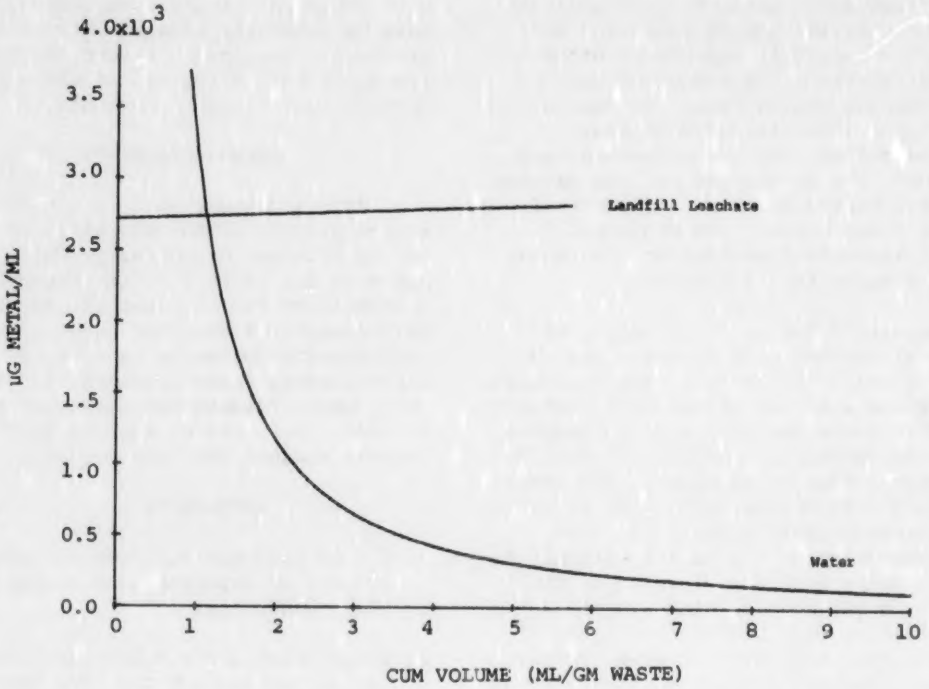

Figure 11. Comparison of Continuous Leaching of Cadmium from Ni-Cd Battery Waste by Landfill Leachate and Water

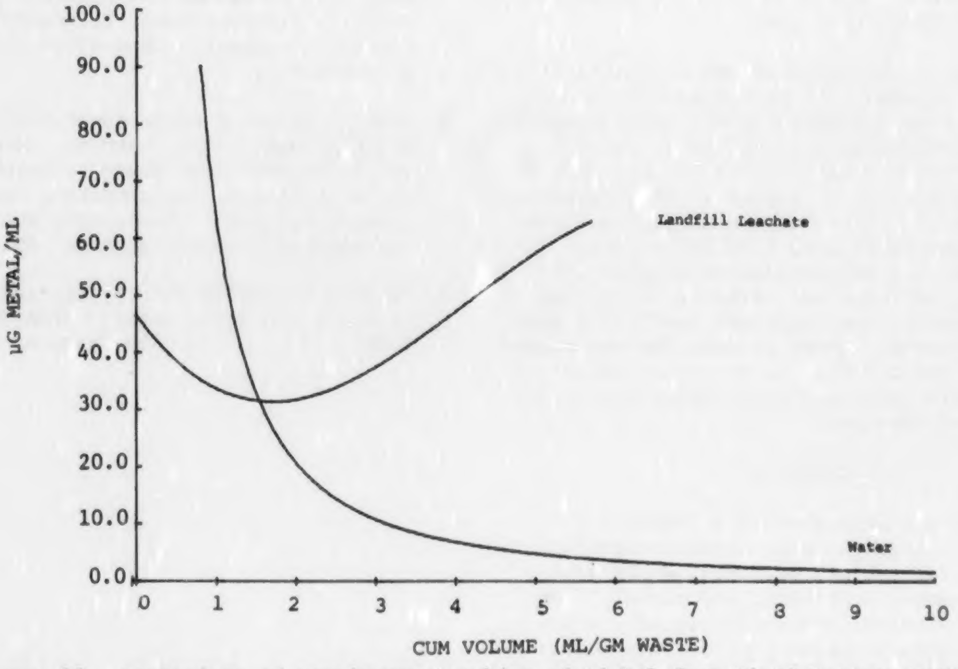

Figure 12. Comparison of Continuous Leaching of Nickel from Ni-Cd Battery Waste by Landfill Leachate and Water

In the first batch extract, approximately 6.5 percent of the solids dissolved in water but very little cadmium or nickel appeared; instead, the dissolved solids were primarily excess base. Subsequent batch extractions yielded much less dissolved solids, and as the excess base washed out, the pH dropped and the cadmium and nickel became soluble. As observed with the other wastes, the municipal landfill leachate dissolved more material from this waste than did water.

Because of the high solubility of the nickel-cadmium battery waste and the ease with which both metals were extracted from the waste in the serial batch extractions, this waste was also used to compare water with municipal landfill leachate by continuous column extractions. The waste was packed into columns made from 37 millimeter diameter glass tubing with an 8 millimeter diameter tip on the bottom. A piece of glass wool was placed over the bottom hole and covered with washed quartz sand (Ottawa Sand). One hundred grams of the waste was mixed with an equal weight of Ottawa Sand and packed into the column, occupying a depth of approximately 10 centimeters. (It was mixed with sand so that the desired flow rate of 1.3×10^{-4} centimeters per second could be obtained through this dense waste.)

Figures 11 and 12 are plots of the results of the columnar extraction of cadmium and nickel from the waste by the two solvents. The figures show that the water extracted more of both metals initially than did municipal landfill leachate (the concentration of both metals in the extract was not measured at zero time for water). However, the concentrations dropped off rapidly in the water extracts while they increased in the municipal landfill leachate extracts. These results further demonstrate the increase in potential hazard if industrial and municipal wastes are disposed of together.

CONCLUSIONS

It has been shown that municipal landfill leachate will extract significantly greater amounts of toxic metals (cadmium, copper, chromium, and nickel) from electroplating, inorganic pigment, and nickel-cadmium battery production wastes than will water. It was concluded that if hazardous industrial wastes are codisposed with municipal refuse, the disposal site must be carefully managed because of the potential increase in hazard due to the increased solubilization of these metals by municipal landfill leachate.

ACKNOWLEDGEMENT

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COMPATIBILITY OF LINERS WITH LEACHATE

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ABSTRACT

This paper presents the results of one year of exposure of various liner materials to sanitary landfill leachate. These materials include:

Six admix materials:

- 2 Asphalt concretes
- 1 Soil asphalt
- 2 Asphaltic membranes
- 1 Soil cement

Six primary flexible polymeric liner specimens based upon the following polymers:

- Butyl rubber
- Ethylene propylene rubber (EPDM)
- Chlorinated polyethylene (CPE)
- Chlorosulfonated polyethylene
- Polyethylene (PE)
- Polyvinyl chloride (PVC)

Additional small membrane specimens, including polypropylene, polybutylene, and neoprene, were also exposed and tested.

The first year of exposure did not result in losses of impermeability in any of the liners. There were losses, however, in the compressive strength of the admix liner materials. Also, there were losses in the physical properties of some of the polymeric membranes and swelling of most of these membranes. The seams of several lost strength, with the heat-sealed seams holding up best as a group.

Among the flexible membranes, the crystalline polymers, polyethylene, polypropylene, and polybutylene, sustained the least change during the first year of exposure; however, these membranes are susceptible to easy puncturing and tearing which would cause problems on installation. The thermoplastic membranes, chlorinated polyethylene, chlorosulfonated polyethylene, and polyvinyl chloride, tended to swell the most. The vulcanized rubbery liner materials, e.g. butyl and EPDM, changed little during the exposure period, but had the lowest initial seam strength.

The data presented must be considered as preliminary in an ongoing project; it is premature to estimate the service lives of the various materials or to make relative comparisons among them for use in a given installation without consideration to costs and to the specifics of the installation.

INTRODUCTION

The use of impervious materials to line sanitary landfills appears to be a promising method for intercepting and controlling leachate generated in a fill to prevent it from polluting surface and ground water. Although many liner materials appear to be potentially useful for this purpose, information available regarding the effects of exposure to leachate on them is very limited, even for relatively short exposure times.

The project on which this paper is based was undertaken to assess the effects of landfill leachate on the properties of a broad range of liner materials. They include asphaltic membranes, soil asphalt, soil cement, and flexible polymeric membranes. Soils and clays were specifically excluded from the investigation as they are being covered in other projects.

TECHNICAL APPROACH AND REVIEW

Technical Approach

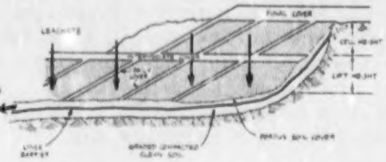
Taking into account the wide diversity in the types of materials that are candidates for lining landfills and the urgent need for information regarding the relative merits of the various liners and their expected lifetimes in a landfill environment (1,2), the following overall approach is being taken:

- Expose liner specimens individually to leachate under pilot conditions that simulate as closely as possible those conditions a liner would encounter at the bottom of a real landfill. The simulated sanitary landfills are designed and constructed to ensure anaerobic conditions, and the leachate generated is representative of the leachate generated in sanitary landfills.
- Select for exposure testing 12 specific liner materials from the various types of liner materials that have been successfully used in lining pits, ponds, lagoons, canals, etc., to prevent seepage of water or various wastes, and that appear suitable for lining sanitary landfills. Soils and clays are excluded from this project as they are covered in other investigations.
- Each test specimen is of sufficient size so that physical tests can be made to measure the effects of exposure to leachate and, if appropriate, a typical seam is incorporated for testing.
- Subject the liner specimens to appropriate tests for the specific type of liner. Measure properties that could be expected to reflect on the performance of the liner in sanitary landfills and determine the changes in properties as a function of exposure time.
- Seal the liner specimens in individual simulated landfills so that whatever seepage might come through can be collected and tested. This required special efforts to avoid leachate by-passing the liner or channeling through the liner, particularly in the cases of soil cement and soil asphalt liners.
- Create equal conditions in all simulated fills, so that valid comparison between liners can be made. To accomplish this, fill the simulated landfills with well-compacted, shredded municipal refuse. Compaction, composition, and amount of refuse are as equal as possible in each of the 24 cells so that a relatively highly concentrated and equal leachate is generated in all the cells.
- After the refuse in the cells is saturated, i.e. brought to "field capacity", generate leachate by adding 1 inch of tap water every 2 weeks (26 inches per year) and allow leachate to pond on the liner at a depth of about 1 foot by draining and collecting leachate every other week.

Environment of a Liner in a Landfill

As with any product, the service life of a liner will depend upon the materials of which it is made and on the environment in which it must function. Some important conditions that exist at the bottom of a landfill which should have major effects on service life are:

- As shown in Figure 1, the liner is generally placed on a relatively firm, smooth surface that has been graded to allow drainage and compacted. Rocks, stumps, and other objects that could cause cracking of hard liners have been removed. Settlement, however, could cause a brittle or weak liner to fail.
- Only modest hydraulic head pressure on the liner exists, since drainage above



CONCEPT OF LINING A SANITARY LANDFILL

FIGURE 1

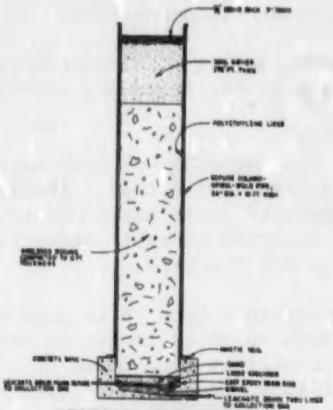
the liner is designed to take place continually. Before refuse is added a porous soil is placed on the liner to protect it from puncture.

- At the bottom of a compacted landfill, anaerobic conditions normally exist, i.e. there is no oxygen present to cause oxidative degradation of the liner.
- The liners are in total darkness; there is no ultraviolet light which can degrade many organic and polymeric materials.
- The conditions are wet, particularly if leachate is being generated regularly, and could result in the leaching of ingredients, e.g. the plasticizer in PVC, from a liner compound.
- The temperature at the bottom of a landfill is usually cool, in the range of 40 to 70°F, which is favorable for extended service life. High temperature can be generated within the fill if air is present, a condition which exists when refuse is first placed or is not compacted.
- Acidic conditions due to the leachate may deteriorate some liners, e.g. soils and soil cement.
- High saline concentrations in leachate may exchange with clays and soils to increase permeability.
- Dissolved organic constitutents in the leachate may degrade some of the liners made of organic materials, e.g., polymer membranes and asphaltic liners.

The effects of these environmental conditions will differ on the various barrier materials. However, it appears that mechanical failure during installation or during operation of the fill due to settling of the soil is the most probably source of failure of a liner in a landfill.

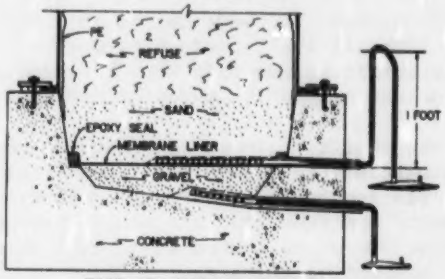
Simulated Landfills

The leachate generator and exposure cell, shown in Figures 2 and 3, was designed (1,2) to create conditions which closely simulate the environment in a landfill.



LEACHATE GENERATOR

FIGURE 2



BASE OF LEACHATE GENERATOR

FIGURE 3

Twenty-four of these simulated landfills have been constructed at the Sanitary Engineering Research Laboratory of the University of California, Berkeley. The site, at the Richmond Field Station of the University, on San Francisco Bay, has a moderate and uniform temperature over the entire year, mostly in the 55 to 60°F range.

The 12 liner materials were mounted in duplicate in the 24 generators. 12 of these generators were dismantled and the liners removed and tested after 12 months of exposure to leachate and the remaining 12 will be dismantled after 42 months of exposure.

Selection of Specific Materials for Exposure Testing

The selection of specific liner materials for testing as barriers in the simulated landfills involved two major steps:

1. Selecting a type of material and polymer for the admix liners and flexible membranes.

2. Within each type, selecting a specific material or composition.

In making this final selection of specimens to test the following factors were considered:

1. The thickness of each material selected was typical of that normally employed except that, if there was a choice, thin liners were selected so as to accelerate the effects of the leachate.

2. Within a type, high quality compositions were selected, e.g., in selecting specific membranes from those of a given type available, the membrane with best physical properties was generally selected.

No attempt was made to obtain liner samples from all liner producers, but to select specific liners that were representative of the respective types of materials.

The specific liners selected and mounted as barriers in the bases of the simulated landfills include six flexible synthetic polymeric membranes and six admix liner materials. They are listed in Table 1 with their respective thicknesses.

TABLE 1. LINER MATERIALS SELECTED FOR LEACHATE EXPOSURE TESTS

Material	Thickness
Polymeric liner membranes:	mils
Polyethylene (PE)	10
Polyvinyl chloride (PVC)	20
Butyl rubber	63
Chlorosulfonated polyethylene, with fabric reinforcement	34
Ethylene propylene rubber (EPDM)	51
Chlorinated polyethylene (CPE)	32
Admix materials	inches
Paving asphalt concrete	2.2
Hydraulic asphalt concrete	2.4
Soil cement	4.5
Soil asphalt	4.0
Bituminous seal	0.3
Emulsion asphalt on fabric	0.3

In addition to the primary membrane specimens, 42 small secondary polymeric specimens were buried in the sand for exposure testing. They included membranes of several additional polymers which are potentially useful as liners, i.e., neoprene, polypropylene, and polybutylene.

Refuse Characteristics

Approximately 12 tons of residential refuse were ground over a period of a week in an Eidal Mini-Mill Grinder (Model 100) without classification. The shredded refuse was delivered to the site in 3 loads and was systematically loaded into the 24 generators on a rotating basis.

About 950 pounds of refuse, having a water content of 12 to 15%, were added to each of the generators in 45 to 47 loads or lifts. This amount is equivalent to 1150 pounds of refuse per generator of 30% water content, or about 1240 pounds of refuse per cubic yard at 30% water content.

Table 2 shows the composition of the composite sample:

TABLE 2. COMPOSITION OF REFUSE

Material	Percent
Water	12.2
Paper	53.6
Cloth	0.8
Plastics and rubber	4.9
Wood, garden and food waste	5.1
Oils and fats	0.9
Metal	7.6
Glass, rock, and soil	14.9
Total	100.0

The content of identifiable food waste was low. It is assumed that the refuse must have been collected from a neighborhood where most food wastes are flushed into the sewers through sink disposal units. The actual content of putrescible organic material was somewhat higher than shown, as some pieces of paper and plastics in the larger size fractions were obviously saturated with fats, blood, etc.

MONITORING THE GENERATORS (3)

During the first year of exposure of the liners to leachate (November 1974 - November 1975), monitoring the generators consisted of the following:

1. Adding 2 gallons of tap water to each generator on a biweekly basis. This amount equals 1 inch of water per 2 weeks or 26 inches per year.

2. Collecting leachate on a biweekly basis.

3. Measuring biweekly the ambient temperature and the temperatures in the refuse of 4 of the generators.

4. Analyzing, on a 4-week basis, the leachate for the the following:

- Chemical oxygen demand (COD)
- pH
- Total solids
- Volatile solids
- Total volatile acids as acetic acid (This was discontinued after 2 months and analyses were made of the individual volatile acids.)

The temperatures observed in the refuse of the 4 generators very closely equalled ambient temperature (10 to 20°C). Temperatures in the refuse exceeding ambient were only observed during the first few days after the cells were loaded with refuse. By the time the thermocouples had been placed in the cells, the temperatures within the cells had already dropped essentially to ambient temperature.

During the first year the method of collecting the leachate was changed. Initially, the leachate was allowed to accumulate in the cells and then drained. Consequently, the leachate ponded on the liners at various heights, although efforts were made by intermediate drainings to keep the height between 1 and 2 feet. Later, "U" tubes were installed at a height of 1 foot above the liner surface and the leachate was drained continually, leaving a head of 1 foot on the liners at all times. In making this change the collection bags were changed from polyethylene to polybutylene because of the superior seams which could be obtained by heat-sealing polybutylene. The polyethylene bags failed at the heat-sealed seams when kept under constant stress and continuous draining could not be performed with these bags.

It was recognized early that a large amount of organic acids was being generated in the anaerobic decomposition of the refuse. Several of these organic acids can interact with organic materials such as the membrane and asphaltic liners in the study; butyric acids in particular have adverse effects on many rubbery and plastic materials. Consequently, analyses were made for individual organic acids, as shown in the analyses made prior to dismantling the generators to recover the liners exposed for one year to the leachate (Table 3). Results of Breland (4), who reported data on individual organic acids, are also shown.

TABLE 3. ANALYSIS OF LEACHATE

		Breland (4)
Ambient temperature, °C	14	--
Total solids, %	3.31	1.25
Volatile solids	1.95	--
Nonvolatile solids	1.36	--
COD, g/l	45.9	18.0
pH	5.05	5.1
TVA, g/l	24.33	9.3
Organic acids		
Acetic, g/l	11.25	5.16
Propionic, g/l	2.87	2.84
Isobutyric, g/l	0.81	--
Butyric, g/l	6.93	1.83
Consolidation of refuse at 1 year, %	6	--

During the first year of monitoring the cells, the leachate entered the bases below the liners in only 3 of the cells. 2 of these liners, soil asphalt and paving asphalt concrete, leaked whereas the leakage in the third was caused by a failure of the epoxy sealing compound around the periphery of the specimen.

DISASSEMBLY OF GENERATORS AND RECOVERY OF LINERS (3)

Twelve of the 24 leachate generators and exposure cells containing the test liner specimens were dismantled in November 1975 and the liner specimens recovered for laboratory testing. This was done 52 weeks after the refuse in the columns had been brought to field capacity and leachate began ponding on the liners.

The major problem faced in dismantling the generators and recovering the exposed liners was to perform the operations without damaging the liner specimens. The individual filled columns weighed approximately 3000 pounds.

When the generators were dismantled, the refuse was inspected and a photographic record made. The appearance of the refuse showed that it had deteriorated very little during the course of the year. Pieces of newspaper could be read and colors were retained in both paper and pieces of fabric. Organic material, leaves, twigs, etc., also showed little damage. Pieces of plastic and metal (aluminum, tin cans, pennies, etc.) were little changed. However, pieces of rubber, such as rubber bands, were highly swollen and some pieces of what appeared to be polyvinyl chloride, such as used in wallets, had become extremely hard.

The moisture content of the refuse taken from the generators was found to be about 60%.

RESULTS OF ONE YEAR OF EXPOSURE OF LINERS TO LEACHATE

Admixed Materials

The asphalt concrete and soil asphalt liners lost drastically in their compressive strength; however, they maintained their impermeability to leachate. The asphalt binder, which normally hardens on aging in air, became softer indicating possible absorption of organic components from the leachate.

The soil cement lost some of its compressive strength; however, it hardened considerably during the exposure period and cored like a portland cement concrete. It became more impermeable during the exposure.

Inhomogeneities in the admix materials, which probably caused the leakage in the paving asphalt and soil asphalt liners, indicate the need for considerably thicker materials in practice. Thicknesses of 2 to 4 inches were selected for this experiment to give an accelerated test and were designed with an appropriately sized aggregate. The same compositions in the second set of 12 liners have not leaked after 27 months of exposure to leachate.

The asphalt membranes withstood the leachate for 1 year, although they did swell slightly. There was no indication of disintegration or dissolving of the asphalt.

Polymeric Membranes

The changes in the physical properties of the polymeric membranes during the first year of exposure to leachate are presented in Table 4. Overall the change in the physical properties of the membranes was relatively minor. They all tended to soften, probably due to the absorption of leachate. On the other hand, there was a substantial loss in seam strength in the polyvinyl chloride, the chlorosulfonated polyethylene, and the chlorinated polyethylene liners. The seam strength of the butyl and EPDM liners decreased less, but they had lower strength prior to exposure. The polyethylene maintained the highest seam strength reflecting the fact that it was heat-sealed.

Of the 6 polymeric membranes, the polyethylene film best maintained overall properties during the exposure period. It also absorbed the least amount of leachate. However, this liner material has low puncture resistance. The butyl and EPDM liners changed somewhat more in physical properties than did the polyethylene during the exposure period. In particular, they maintained their stress-strain properties and did not soften; they retained their respective seam strengths, but their original values were low. The 3 remaining membranes, PVC, chlorosulfonated polyethylene, and CPE, were about equal; they all tended to soften and lose in hardness, tensile properties and in seam strengths, even though they had good initial values. These latter materials are all thermoplastic and unvulcanized.

The fact that no leachate appeared below the membrane liners during the year indicates high impermeability. No test of water permeability was made of the exposed specimens; samples have been retained for possible future tests. On the other hand, related closely to the water permeability of materials is their absorption of water. The absorption can thus result in significant swelling of a liner and a significant increase in its permeability to water and possibly to dissolved components.

TABLE 4. EFFECT ON THE PROPERTIES OF POLYMERIC MEMBRANE LINERS OF 1 YEAR OF EXPOSURE TO LEACHATE FROM SIMULATED SANITARY LANDFILLS
(Data in U.S. Customary Units)

Item	Test method	Exposure time, years	Polyethylene	Polyvinyl chloride	Butyl rubber	Chloro-sulfonated polyethylene	Ethylene propylene rubber	Chlorinated polyethylene
Liner number	--	--	21	17	7	6	16	12
Generator number	--	--	19	20	21	22	23	24
Thickness, mils	--	0	11-12	20-21	61-65	32-36	49-53	31-32
	--	1	11	21	64	38	51	35
Tensile strength, psi	ASTM D412	0	2145	2580	1435	1765	1475	2270
		1	2465	2350	1395	1640	1455	1810
Elongation at break, %	ASTM D412	0	505	280	395	250	410	410
		1	560	330	410	300	435	400
Tensile set, %	ASTM D412	0	422	73	17	111	16	429
		1	432	57	14	106	12	208
S-200[a], psi	ASTM D412	0	1260	1965	690	1520	760	1330
		1	1205	1550	685	1245	740	1090
Tear strength (Die C), ppi	ASTM D624	0	390	335	180	300	181	255
		1	496	450	202	305	195	320
Hardness (Duro A - 10 sec.)	ASTM D2240	0	98	76	51	79	54	85
		1	--	64	50.5	64	51.5	65.5
Puncture resistance[b]	Fed. Std. 101B, Method 2065							
Force, lb		0	13.9	25.8	44.8	32.9	39.4	47.0
		1	14.8	30.1	49.5	57.0	40.1	49.8
Elongation, in.		0	0.76	0.69	1.22	0.60	1.44	1.04
		1	0.80	0.70	1.20	0.88	1.18	0.98
Volatiles at 105°C, %	--	1	0.02	3.55	2.02	12.76	5.54	6.84
Seams								
Method of bonding	--	--	Heat	Cement	Cement (LTV)[c]	Cement	Cement (LTV)[c]	Solvent
Peel strength, ppi	ASTM D413	0	>15.6[d]	4.0	3.8	>30.0[d]	2.5	10.0
		1	>10.3[d,e]	5.1	2.9	3.4	2.0	5.1
Shear strength, ppi	--	0	>20.2[d]	>37.2[d]	30.0	>50.0[d]	14.5	>57[d]
		1	>11.4[d]	>25.6[d]	42.0	40.2	24.3	>35[d]

[a] Stress at 200% elongation

[b] Rate of penetration of probe: 20 inches per minute.

[c] Low temperature curing cement.

[d] Break in the specimen outside of seam.

[e] Seam in the polyethylene liner used in the steel columns; tabs in the liner specimens mounted in base were too short.

In Table 5 the water and leachate absorptions by various polymeric liner materials are presented. The data include the water absorption after 1 year in water, and leachate absorption after 1 year of exposure. The materials which have shown the lowest amount of swell are polyethylene, polybutylene, and polypropylene, in which cases the absorptions are a few tenths of a percent, with the leachate being absorbed slightly more than water. These results would predict the low water transmission which was observed for the polyethylene (3).

TENTATIVE CONCLUSIONS AFTER ONE YEAR OF EXPOSURE

One year of exposure to sanitary landfill leachate resulted in relatively minor deterioration to most of the liner materials which are being evaluated. In no case was there a significant increase in water permeability. Admixed materials, such as soil asphalt and asphaltic concretes, lost significantly in compressive strength, but did not increase in permeability. The polymeric membranes swelled and softened to various degrees but did not become permeable. These results indicate that substantially longer exposures would be required to be able to determine whether the liners would become more permeable when in contact with leachate.

The small amount of change which took place in the properties of the liners after one year of exposure indicates that an additional year would have a minor effect upon properties. Therefore, the liners which remain in the test should be exposed considerably longer than the additional year before being retrieved and tested. The swelling of most of the polymeric materials by the leachate would indicate a possible

TABLE 5. WATER AND LEACHATE ABSORPTION BY POLYMERIC LINERS
(Data in percent absorbed by weight)

	Liner no.	Water-RT 1 year	Leachate 1 year
Butyl rubber	7[a]	1.60	1.78
	22	1.70	2.32
	24	1.10	1.0
Chlorinated polyethylene (CPE)	12[a]	13.10	9.0
	13S[b]	19.60	12.4
	23	15.50	10.3
Chlorosulfonated polyethylene	3	17.40	20.0
	4S	18.00	19.0
	6S[a]	9.20	13.64
	14S	11.20	8.71
Ethylene propylene rubber (EPDM)	8	1.40	5.95
	16[a]	4.80	5.50
	18	--	--
	25	1.50	5.59
	26	1.60	8.99
Neoprene	9	22.7	8.73
Polybutylene	20	0.25	0.33
Polyethylene	21[a]	0.20	0.25
Polypropylene	27	0.28	0.40
Polyvinyl chloride	10	1.85	6.72
	11	1.85	5.0
	15	2.10	4.64
	17[a]	1.85	3.29
	19	0.60	0.75

[a] Liners mounted in generator bases.
[b] S = fabric supported liner.

loss in permeability over the longer exposures. Many polymeric materials become increasingly permeable on absorption of the medium with which they are in contact. The testing of permeability of swollen membranes should be undertaken.

The variation in the swelling of polymeric liners, particularly the polyvinyl chloride, indicates a variation in composition which could be a significant factor in the long term performance of a given liner. It is concluded that a study should be included of the composition of these liner materials to determine the effect upon exposure to leachate.

The method being followed in this project to assess the various liner materials is costly and time consuming and certainly is not amenable to specifications requirements. A more simple and more direct test should be developed.

This project will supply information regarding the effects of leachate on various liner materials on a relatively small scale. Correlation of these results with actual landfills is desired in order to assess fully the performance of liner materials.

PLANS AND CURRENT WORK

In view of the above conclusions, the project was modified to accomplish the following:

1. To extend the length of time that the remaining liner materials now being exposed to leachate from 24 months to 42 months. The properties of the liner

materials in the test will thus be tested after 1 year and 3.5 years of exposure to landfill leachate.

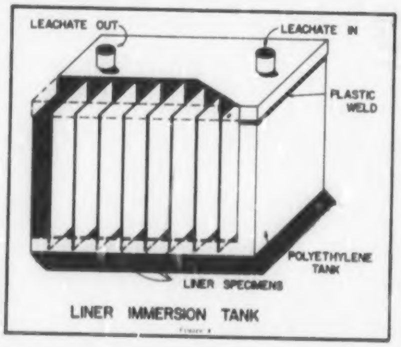
2. To run supplementary exposure tests of various liner materials including several new materials, e.g. elasticized polyolefin, polyester elastomer, unvulcanized EPDM, etc., by immersion in leachate generated and collected from the remaining 12 simulated sanitary landfills.

3. To analyze the basic compositions of the various polymeric membranes for possible use in establishing specifications.

4. To determine the permeability of polymeric membrane liners to water and leachate when swollen as well as when unexposed.

5. To develop simple methods for evaluating materials for lining landfills which could be used in setting up specifications.

Small specimens of a range of polymeric liners are being immersed in the leachate for up to 1.5 years in tanks of polyethylene, such as shown in Figure 4, under anaerobic conditions. These tanks are arranged in series with the leachate from the generators combined and allowed to stream slowly through the tanks. The specimens are large enough for the tests used in the primary evaluation, except for the seam adhesion. Specimens will be removed and tested at intervals.

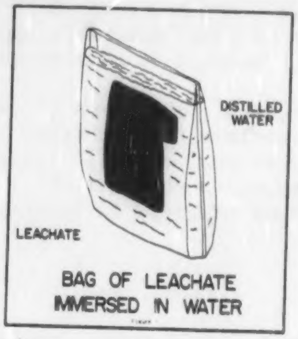


As the permeability of a liner to water and leachate is of primary importance in its performance as an impervious barrier at the bottom of a landfill, several test methods are being used to determine this property particularly of flexible membrane liners. These methods include:

1. ASTM E96, Method BW, to determine the moisture vapor transmission of liners.

2. Tests in a top-pressure permeameter in which approximately 4 atmospheres of pressure are being applied to specimens, both as received and after swelling in water at 70°C.

3. Immersion in distilled water of sealed bags prepared from the flexible liner materials and containing leachate. In this test the pH and conductivity of the distilled water are being monitored and the weight of the bags is being measured at intervals (Figure 5).



The primary exposure cells continue to be monitored. There is a gradual change in composition of the leachate; both volatile and nonvolatile solids are decreasing, as of February 1977, as are COD and organic acids. The latter may be a reflection of the lower ambient temperature. The refuse has consolidated 11% in the 27 months since it was brought to field capacity in December 1974.

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PREDICTING CADMIUM MOVEMENT THROUGH SOIL AS INFLUENCED BY LEACHATE CHARACTERISTICS

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ABSTRACT

Cadmium movement through soil can be expressed as a function of the physical components of the system. An analytical solution to a partial differential equation was fitted to data from soil-column experiments. The resulting coefficients are regressed against readily measurable properties of the leachate. Results indicate that these coefficients (corresponding to travel rates of cadmium through soils) are highly related to both total organic carbon content and total soluble ions of the carrier fluid. Migration rates of Cd through different soils are tabulated with relevant characteristics of the leachate, as a simplified, user-oriented predictive tool.

INTRODUCTION

To develop confident guidelines for disposal of potentially hazardous metal-containing wastes, a knowledge of a multitude of microenvironmental factors influencing metal migration rates is necessary. Research identifying some soil and leachate factors has been published by Fuller et al. (1976a,b), and Korte et al. (1976) report metal movement as influenced by soil physical and chemical properties. Leachates of varying properties affect the movement of certain metal ions through clay mineral systems according to Griffin et al. (1976) and Folsom et al. (1976). This paper attempts to investigate the effect of interaction of soils and leachate properties on Cd^{++} movement. Fuller (1977) suggested that the major solution properties of the leachate affecting metal attenuation are (a) total organic carbon content (TOC) and (b) total soluble ion (ION) concentration. Predictions of Cd movement are based on a regression analysis because models based on chemical reactions have rarely been successful.

THE MODEL

The model assumes Cd^{++} movement in soils is described by the Lapidus and Amundson (1952) model. The theoretical curves approximately describing Cd transport in soil columns, under saturated steady flow conditions are matched with actual data. Because of the nonlinearity of the mathematical description, a slightly different approach to parameter estimation is taken. The results of the parameter estimation procedure are regressed with readily measurable soil-leachate system properties. Each of these coefficients correspond to a particular migration rate for Cd. To facilitate management use, the results of the regression analysis and rate calculations are tabulated.

The equation used for describing Cd transport, under saturated steady flow conditions, is

$$D \frac{\partial^2 c}{\partial z^2} = V \frac{\partial c}{\partial z} + \frac{\partial c}{\partial t} + \frac{1}{\alpha} \frac{\partial n}{\partial t} \qquad (1)$$

where

c = Cd concentration in solution ($\mu g/cm^3$)

n = amount of Cd sorbed per unit volume of soil ($\mu g/cm^3$)
D = dispersion coefficient (cm^2/day)
V = pore water velocity (cm/day)
t = time (day)
z = distance in soil (cm)
α = fractional void volume in soil

The $\frac{\partial n}{\partial t}$ term does not include plant uptake, microbial degradation or chemical transformations. The mathematical description of the absorption mechanism is an empirical one. The mechanism may be approximated as a first order kinetic reaction by

$$\frac{\partial n}{\partial t} = K_1 c - K_2 n \qquad (2)$$

The parameters K_1 and K_2 are forward and backward reaction rate constants (day^{-1}), respectively.

The additional relations:

$$c = C_0 \quad , \text{ when } z = 0,\ t > 0 \qquad (3a)$$

$$\left.\begin{matrix} c = 0 \\ n = 0 \end{matrix}\right\} \quad , \text{ when } t = 0,\ z > 0 \qquad (3b)$$

describe input concentrations and initial conditions of the soil.

Equations (1), (2), (3a) and (3b) describe traveling wave movement in semi-finite columns where side influences are ignored. This model and its solution was presented by Lapidus and Amundson (1952). Similar models to describe chemical movement in soils have also been presented by Oddson et al. (1970), and Lindstrom et al. (1971). Davidson and McDougal (1973) noted that the ability of the above models to describe experimental data has not been adequately determined.

The solution to (1) and (2) with boundary conditions (3a) and (3b) as presented by Lapidus and Amundson (1952) is:

$$c(t) = e^{\frac{Vz}{2D}}\left[F(t) + K_2 \int_0^t F(t)dt\right] \qquad (4a)$$

$$\text{where } F(t) = e^{-K_2 t} \int_0^t I_0\left[2\sqrt{\frac{K_1K_2x}{\alpha}(t-x)}\right] \cdot \frac{z}{2\sqrt{D\pi x^3}}\, e^{-\left[\frac{z^2}{4Dx} - xd\right]} dx \qquad (4b)$$

$$\text{and} \qquad d = \frac{V}{4D} + \frac{K_1}{\alpha} - K_2. \qquad (4c)$$

Unfortunately no precise prior knowledge exists for the parameters D, K_1 and K_2. The best estimates of these are obtained by comparisons of calculated curves for various sets of parameters D, K_1 and K_2 with actual data. Curves for finite column lengths are found by restricting z to fixed values.

Since our approach to estimating these parameters is slightly out of the ordinary, we present the steps in some detail.

PARAMETER ESTIMATION

Reactive solute movement in soil columns appears to be variable in nature. Precise replication of data from such experiments is never achieved. Uncontrolled variables such as soil packing, irregular and irreproducible pore geometry, and trapped gases often account for slight variations in experimental breakthrough curves. The first step is therefore measurement of the ability to reproduce data from column experiments.

Replicate column experiments using Ava si.c.l. and leached with 100 ppm Cd in solution with three natural municipal leachates. Table 1 summarizes the error estimates from these studies. In all subsequent column studies the experimental error was assumed to be approximately 3%.

The parameters to be estimated in equation (4a), (4b) and (4c) are D, K_1 and K_2. Theoretical curves are matched with experimental data by use of an iterative nonlinear least squares procedure (Davidon, 1959). The sum of squares may be thought as a measure of the difference between theoretical breakthrough curves and experimental data. This can easily be translated to an average distance, $\bar{d}$, between each experimental and calculated data point. $\bar{d}$ represents a 3-dimensional hyperplane in the parameter space. Points below this plane on the sum of squares surface all give rise to breakthrough curves which match the data to within the experimental error. The projection of these points onto the (K_2,K_1)-plane appears to be a long, narrow region that can be approximated by a line segment.

For example, assume the experimental error is 3.0% and also that 15 data points are given. [Data points were generated by $p(t) = c(t) + \varepsilon$, where $c(t)$ is given by equation (4) with $z = 10$ cm, $V = 3.6$ cm/day, $\alpha = 0.474$, $D = 2.0$ cm^2/day, $K_1 = 7.0$ day^{-1}, $K_2 = 15.0$ day^{-1} and $\varepsilon \sim N(0.0, 0.5)$]. As discussed previously, the data points are satisfactorily approximated by a breakthrough curve whenever the average distance between the two is less than 3.0%. Figure 1 is a rough approximation of a small portion of the sum of squares surface for this example. It is possible to approximate the bottom portion of this surface with a straight line segment. Figure 2 is a graph of a few points on the bottom of this surface and the best fitting line. Figure 3 shows breakthrough curves calculated from parameters lying on the bottom of the surface in Figure 1. These curves all approximate the data points within the experimental error.

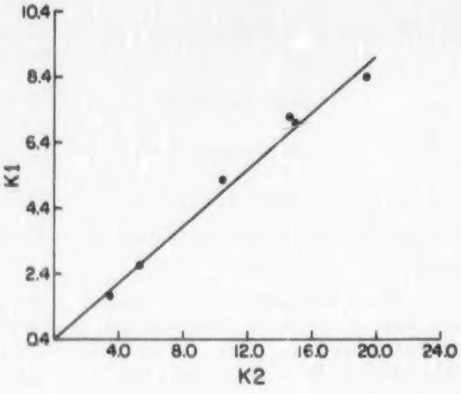

Figure 2. Linear approximation to points of least height on the bottom of Figure 1 showing K_1, K_2 relationship.

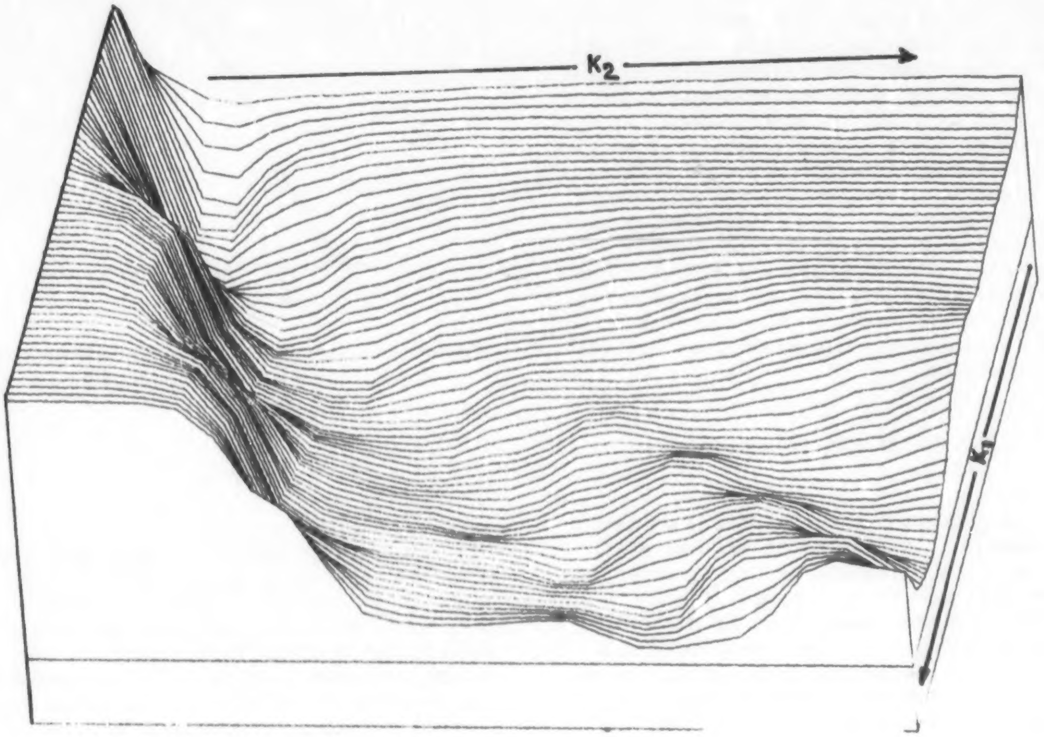

Figure 1. Small section of sum of squares surface for $D = 2.0$ cm^2/day.

TABLE 1. ESTIMATES OF REPRODUCIBILITY FOR SOIL COLUMN EXPERIMENT

Column	Soil Type	Flux (Av.) cm/day	Leachate Source	No. of Replicates	Average Reducibility ppm
1	Ava si.c.l.	5.9	New leachate (II)	4	3.3
2	Ava si.c.l.	7.2	25% of new (II)	3	5.7
3	Ava si.c.l.	6.7	Old leachate (I)	3	3.2

The location of the global minimum on the sum of squares surface followed no definite trend. The above procedure was introduced to alleviate this problem. Moreover every column experiment generates a unique line segment. These segments have a definite trend; fast migration rates have much smaller slopes than slower migration rates.

The nonlinearity of the parameters K_1 and K_2 in equations (4a), (4b), and (4c) prevents a direct translation of the slope, K_1/K_2, into flow rates. This problem is solved by regressing the approximate endpoints of the line segments with TOC and ION. The dispersion coefficient, D, generally increased as the absolute value of the pair of reaction coefficients (K_1,K_2), increased along a line segment. A single dispersion coefficient for each soil is obtained by averaging all the calculated coefficients along each line segment and tabulating an average for all these segments. The flow rates for each slope are then calculated as the average of the flow rates for each of the endpoints. This approach greatly simplified the table construction.

EXPERIMENTAL DESIGN

A factorial experiment was designed to measure the effects of various attributes of the leachate on Cd migration. Each soil was to be leached with Cd-spiked leachate under four levels of ION (inorganic ions). If a particular factor of the leachate significantly influences the migration rate for Cd, it will also cause a variation in the slopes of the calculated line segment for differing values of this factor. These factors are regressed with the slopes of the line segments. Figure 4 is an illustration of the variation in line segment slopes for some actual soils, while Figure 5 shows a variation in slopes with selected leachates.

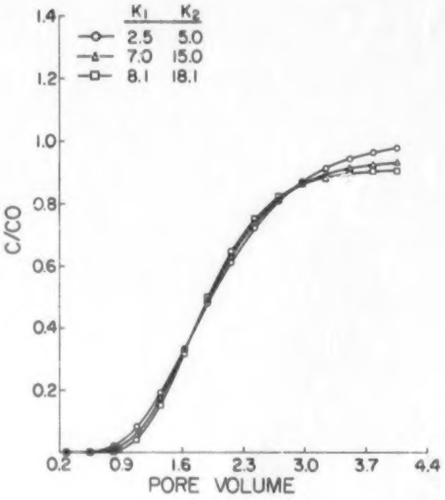

Figure 3. Calculated breakthrough curves for single line segment, D = 2.0 cm^2/day.

PROCEDURE

The soil column procedures have been described by Fuller et al. (1976a) and Korte et al. (1976). In brief, 10-cm lengths of 5 cm PVC pipe were uniformly packed with soil to form a vertical soil column of known density. Soils representing major soil orders were collected from "benchmark" locations through the USA. Natural municipal landfill leachate "spiked" with 100 ppm Cd was displaced through the soil columns under strict,

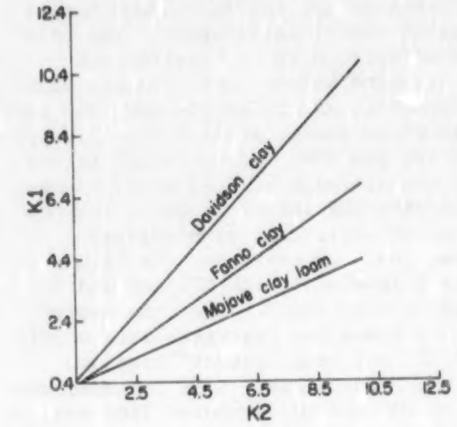

Figure 4. Variation in line slopes as influenced by three soils.

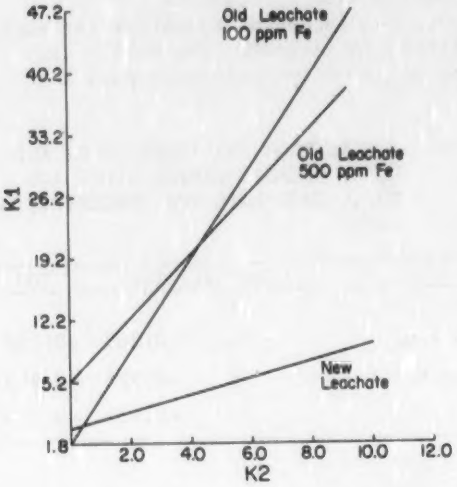

Figure 5. Variation in line slopes as influenced by leachates in Anthony sandy loam.

saturated, anaerobic conditions at known fluxes controlled by use of precision peristaltic pumps. Influent and effluent displacements caught in a fraction collector were analyzed for TOC, inorganic salt, Cd, Fe by standard procedures previously described (Fuller et al., 1976, and Korte et al. 1976a).

Two different sources of leachate were used. One, 3-4-year old solution of low constituents concentration (Fuller et al., 1976) and 1½-2-year old of relatively high constituent concentration (TOC, EC, Fe, etc.)(Fuller et al., 1976).

The latter leachate was diluted to 0, 25 and 50% of original concentration with deionized water as a necessity to evaluate effects of TOC, and inorganic ion concentrations on Cd migration rates through soils.

The pore volumes of Wagram loamy sand (low in clay, 4%), Anthony sandy loam (intermediate in clay, 15%), and Fanno clay (relatively high in clay, 46%) were displaced with four leachates having four levels of TOC of 0.01, 0.18, 0.50, and 1.0% until breakthrough of Cd occurred (i.e., C/Co = 1) or until 20 pore volume was collected. This same procedure was followed for evaluating the effect of Fe levels (100, 500, and 1,000 ppm) and inorganic levels (342, 1,013, and 2,163 ppm) on Cd attenuation at different TOC levels. The varying iron levels were obtained by "spiking" the leachate with desired concentration of FeCl, whereas the salt levels were established by "spiking" each leachate used with various salts ($CaCl_2$, $Mg(NO_3)_2$, NaCl, and KCl) to give the desired concentration at the same cation ratio as the natural landfill leachate.

RESULTS AND DISCUSSION

Line segments calculated from column experiments were regressed against both TOC and ION. The competitive ion term is assumed to be the sum of the soluble ions in solution, Fe, Mn, Ca, Na, and K. High correlations were reached with both of these terms on Anthony sandy loam and Fanno clay. Wagram loamy sand did not show high correlations with either property. However, this may be expected. Wagram loamy sand contained only 4% clay-sized particles.

Table 2 is a summary of the correlation coefficients for the slope of the line segment against the terms TOC and ION. The negative quantities indicate Cd transport rates increase as TOC and ION increase.

The leachate's influence on Cd transport is discussed separately for each soil listed in Table 2. The soil's clay content is given for reference purposes.

TABLE 2. CORRELATION FOR THREE SOILS AND TOTAL ORGANIC CARBON (TOC) AND TOTAL INORGANIC ION CONCENTRATION (ION)

Soil	Clay-<2μ %	TOC	ION
Wagram l.s.	4	-0.203	-0.150
Anthony s.l.	15	-0.928	-0.922
Fanno c.	46	-0.903	-0.994

Tables 3 through 8 summarize the information obtained to date and indicate the approach that will be used to bridge the gap between the simulation model and field application. A regression equation for the slope (K_1/K_2) of the line segment is developed in the form:

$$\text{Slope} = (D \times \text{Factor}) + (E \times \text{Factor}) + \text{----} \text{---} + H$$

where the letters D, E, --- H are numerical coefficients and the Factors are measurable properties of the leachate (eg. TOC and ION). The users will insert the factor information for a particular situation to arrive at a value for slope applicable to their situation. The tables list contaminant movement rates for each slope and relative concentration (C/C_0). Using Table 3 as an example, suppose a user has a waste that is releasing 2.0 ppm cadmium (C_o) in its leachate and 0.4 ppm cadmium (C) is the concentration limit in the soil solution below the disposal site. Then the user is concerned about the relative concentration 0.2 (C/C_o = 0.4 ppm/2.0 ppm = 0.2). Supposing, additionally, that using measured properties of the leachate the user has calculated the slope to be 0.12, then Table 3 shows that the migration rate of the cadmium will be about 33.7 cm/day. If the critical concentration limit in the soil solution was, instead, 1.2 ppm, then the relative concentration would be 0.6 and the migration rate would be 26.7 cm/day.

Wagram loamy sand: The low correlations may be expected for Wagram l.s. since it is a very loose, highly siliceous sand (88%) with only 4% secondary clay mineral. For widely varying leachates there were only slight differences in observed breakthrough curves. Some changes were within experimental error. The only notable difference in Cd breakthrough curves occurred between the leachate with 13,010 ppm TOC, and 2,762 ppm ION, and the leachate whose characteristics are 116 ppm TOC and 428 ppm ION. Cd under the former reached breakthrough at 2.43 pore volumes, while Cd with the latter leachate reached breakthrough after 3.56 pore volumes. These two leachates represent the highest and lowest levels of both TOC and ION for a Wagram carrier fluid. The line segment slopes for these two leachates were 0.1010 and 0.1250, the more "potent" leachate having the lower slope. Some intermediate levels of TOC and ION produced line segment slopes which were outside the range given by the highest and lowest levels.

An initial estimate of the relationship between slope with TOC and ION concentration is:

$$\text{Slope} \doteq -9.48\text{x}10^{-6}\text{x}(\text{TOC ppm}) - 6.36\text{x}10^{-5}\text{x}(\text{ION ppm}) + 0.15 \qquad (5)$$

Tables 3 and 4 are tabulated flow rates for various values of slope given by equation (5) for solution speeds of 37.40 cm/day (fast) and 9.35 cm/day (slow), respectively. The dispersion coefficient for these tables is 19.50 cm^2/day, and the fractional void volume is 0.350. The range of tables 3 and 4 encompasses all observed slopes in the experiments.

Both equations (5) and Tables 3 and 4 indicated Cd movement in Wagram l.s. is extremely rapid. Furthermore Cd can move through saturated Wagram soil at velocities higher than the carrier fluid. For a slope of 0.01 the leading edge travels at approximately 42.66 cm/day with a carrier fluid velocity of 37.40 cm/day. This phenomenon is also pronounced at lower velocities (see Table 4). It should be noted that these velocities are calculated from the total volume of solute per day divided by the void volume per centimeter of soil column.

Anthony sandy loam: Cd movement in this soil changed radically with leachate characteristics. For leachate components of 10,510 ppm TOC and 2,300 ppm ION, Cd breakthrough was recorded at 6.53 pore volumes. Only 60% of initial concentration was measured in the column effluent at 25.04 pore volumes for a leachate whose characteristics are 105 ppm TOC and 398 ppm ION. At low levels of TOC, the ION effect was most noticeable. For instance, 60% of initial Cd concentration was achieved at 14.60 pore volumes with a leachate of 105 ppm TOC and 1,398 ppm ION.

Slope of the computed line segment also changed with leachate characteristics. The result of the regression analysis for Anthony s.l. is:

$$\text{Slope} \doteq -1.67\times10^{-4}\times(\text{TOC ppm}) - 7.66\times10^{-4}\times(\text{ION ppm}) + 4.59 \qquad (6)$$

Tables 5 and 6 are tabulated Cd transport rates for corresponding slopes at flow rates of 10.047 cm/day (slow) and 40.19 cm/day (fast), respectively. These tables span the range of observed line segment slopes from experimental data. The dispersion coefficient and fractional void volume for both of these tables are 13.58 cm^2/day and 0.333, respectively.

The average slope for column experiments with Anthony s.l. is 2.19. This corresponds to the leading edge travel rates of 1.89 cm/day and 6.62 cm/day in slow and fast carrier fluid rates, respectively. It is interesting to note from Tables 5 and 6 that Cd transport rates are lower than carrier fluid rates for all levels of leachate constituents. Table entries of 0.0 indicate that particular concentration of Cd calculated from equation (4a) was not reached in 45 days. This may be due to insufficient accuracy in the numerical integration.

Fanno clay: Transport of Cd in various leachates through Fanno c. behaved in a similar manner to Anthony s.l. However, the flow rates for Cd were appreciably lower than in Anthony. At high levels of TOC the effect of ION concentration was apparent but not as great as at lower levels of TOC.

The result of regression of slope with TOC and ION concentration is:

$$\text{Slope} \doteq -6.22\times10^{-5}\times(\text{TOC ppm}) - 2.03\times10^{-3}\times(\text{ION ppm}) + 6.69 \qquad (7)$$

Tables 7 and 8 tabulate Cd flow rates for various slopes at solution speeds of 6.76 cm/day (slow) and 27.06 cm/day (fast). The coefficient and fractional void volume for both tables are 9.67 cm^2/day and 0.495, respectively.

Some regression difficulty was encountered during the table construction for Fanno c. For this reason these tables were built to include slopes only up to 4.7. Further column experiments are underway to hopefully correct this situation. These tables do include most of the observed line segments to date.

Tables 7 and 8 show Cd movement is considerably slower in Fanno c. than in both Wagram and Anthony s.l.

In Fanno c., Cd travel rates are not greater than carrier fluid rates. The average slope observed for Fanno clay was 3.18. The leading edge travel rate of Cd concentration for this slope is approximately 1.25 cm/day for slow solution speeds and 4.55 cm/day for fast solutions.

CONCLUSIONS

While equations 5, 6, and 7 do adequately approximate present experimental results, insufficient data have been collected for any general predictions of Cd transport in soils and the results presented in this paper should be taken as preliminary. The results to date are consistent with previous work showing the effect of leachate organic content and ionic strength on solute movement in soils. Additionally, it appears that it will be possible to develop simple and effective user-oriented tools for predicting contaminant movement in soils to improve the selection and operation of disposal sites.

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TABLE 3. CADMIUM TRANSPORT RATES (cm/day) IN WAGRAM SAND FOR A SOLUTION SPEED OF 37.40 cm/day.

	C/C_o				
SLOPE OF LINE SEGMENT[1]	0.2	0.4	0.6	0.8	0.9
0.01	42.66	37.92	34.56	31.75	29.68
0.02	42.13	37.30	33.96	30.75	28.44
0.03	40.63	36.11	32.97	29.94	28.44
0.04	39.92	35.55	32.05	29.17	27.08
0.05	39.23	34.47	31.60	27.75	25.85
0.06	38.56	33.96	30.34	27.75	25.85
0.07	37.64	33.24	29.76	26.66	24.61
0.08	36.57	32.40	29.09	26.12	24.61
0.09	36.05	32.00	28.44	25.60	24.61
0.10	35.55	30.84	27.82	25.09	22.85
0.11	34.59	30.11	27.23	24.61	22.85
0.12	33.68	29.76	26.66	24.15	22.06
0.13	33.68	28.76	25.60	23.27	22.06
0.14	32.82	28.44	25.00	22.85	21.33
0.15	32.00	27.82	25.09	22.06	21.33
0.16	31.21	27.23	24.61	22.06	20.64
0.17	31.21	26.66	24.15	21.33	20.00
0.18	30.11	26.59	23.27	21.33	19.69
0.19	29.25	26.25	22.75	20.89	18.96
0.20	29.25	25.60	23.27	20.48	18.96

[1] Slope = $-9.48 \times 10^{-6} \times (\mathrm{TOC}) - 6.36 \times 10^{-5} \times (\mathrm{ION}) + 0.15$

TABLE 4. CADMIUM TRANSPORT RATES (cm/day) IN WAGRAM SAND FOR A SOLUTION SPEED OF 9.35 cm/day

	C/C_o				
SLOPE OF LINE SEGMENT[1]	0.2	0.4	0.6	0.8	0.9
0.01	12.82	10.26	8.55	7.07	6.41
0.02	12.82	10.01	8.37	6.84	6.41
0.03	12.43	9.77	8.04	6.84	6.41
0.04	12.07	9.54	7.89	6.41	5.70
0.05	11.72	9.32	7.74	6.21	5.40
0.06	11.72	9.12	7.60	6.21	5.13
0.07	11.40	8.92	7.46	6.03	5.13
0.08	10.80	8.55	7.32	6.03	5.13
0.09	10.80	8.37	7.07	5.70	5.13
0.10	10.80	8.37	6.95	5.40	5.13
0.11	10.26	8.20	6.84	5.40	5.13
0.12	10.01	8.04	6.61	5.40	5.13
0.13	10.01	7.74	6.61	5.26	4.46
0.14	9.54	7.60	6.41	5.13	4.27
0.15	9.32	7.46	6.21	5.13	4.27
0.16	9.32	7.32	6.03	5.13	4.27
0.17	9.32	7.32	5.86	4.88	4.27
0.18	8.92	7.07	5.70	4.66	4.27
0.19	8.92	6.95	5.70	4.66	4.27
0.20	8.92	6.84	5.54	4.66	4.27

[1] Slope $\doteq -9.48 \times 10^{-6} \times (\mathrm{TOC}) - 6.36 \times 10^{-5} \times (\mathrm{ION}) + 0.15$.

TABLE 5. CADMIUM TRANSPORT RATES (cm/day) FOR ANTHONY s.l. FOR A SOLUTION SPEED OF 10.047 cm/day.

	C/C_o				
SLOPE OF LINE SEGMENT[1]	0.2	0.4	0.6	0.8	0.9
0.3	6.27	5.59	5.05	4.60	4.14
0.6	4.60	3.88	3.35	2.89	2.63
0.9	3.95	2.71	2.14	1.84	1.61
1.2	3.03	2.39	2.02	1.72	1.43
1.5	2.47	2.03	1.72	1.41	1.41
1.8	2.21	1.71	1.43	1.14	1.36
2.1	1.89	1.53	1.34	1.19	1.23
2.4	1.73	1.40	1.17	0.99	0.00
2.7	1.51	1.14	1.01	0.87	0.00
3.0	1.39	1.12	0.94	0.79	0.00
3.3	1.26	1.03	0.88	0.76	0.00
3.6	1.22	0.95	0.81	0.68	0.00
3.9	1.12	0.89	0.68	0.47	0.00
4.2	1.00	0.77	0.60	0.00	0.00
4.5	0.94	0.71	0.58	0.00	0.00
4.8	0.89	0.70	0.50	0.00	0.00
5.1	0.84	0.70	0.50	0.00	0.00
5.4	0.82	0.70	0.50	0.00	0.00
5.7	0.82	0.70	0.42	0.00	0.00
6.0	0.80	0.70	0.42	0.00	0.00

[1] Slope $\doteq -1.67 \times 10^{-4} \times (TOC) - 7.61 \times 10^{-4} \times (ION) + 4.59$

TABLE 6. CADMIUM TRANSPORT RATES (cm/day) IN ANTHONY s.l. FOR A SOLUTION SPEED OF 40.19 cm/day.

		C/C_o				
		0.2	0.4	0.6	0.8	0.9
SLOPE OF LINE SEGMENT[1]	0.3	25.23	22.49	19.89	17.83	16.16
	0.6	18.05	15.36	13.54	11.69	10.72
	0.9	13.70	11.42	10.07	8.78	7.78
	1.2	10.80	9.33	8.20	7.33	6.41
	1.5	9.12	7.74	6.84	6.22	5.70
	1.8	7.60	6.62	5.86	5.26	4.88
	2.1	6.62	5.78	5.26	4.66	4.27
	2.4	5.95	5.13	4.56	4.10	3.80
	2.7	5.26	4.56	4.10	3.73	3.66
	3.0	5.13	4.36	3.73	3.20	3.02
	3.3	4.66	3.95	3.42	2.93	2.70
	3.6	4.30	3.66	3.12	2.71	2.62
	3.9	3.95	3.41	2.87	2.52	2.44
	4.2	3.71	3.10	2.71	2.43	2.51
	4.5	3.45	2.91	2.59	2.47	2.37
	4.8	3.34	2.71	2.53	2.37	2.37
	5.1	2.99	2.58	2.42	2.37	2.19
	5.4	2.84	2.47	2.37	2.19	1.89
	5.7	2.64	2.37	2.23	1.96	1.49
	6.0	2.53	2.37	2.14	1.67	1.19

[1] Slope $\doteq - 1.67 \times 10^{-4} \times (\text{TOC}) - 7.61 \times 10^{-4} \times (\text{ION}) + 4.59$

TABLE 7. CADMIUM TRANSPORT RATES (cm/day) IN FANNO c. FOR A SOLUTION SPEED OF 6.76 cm/day.

	C/C_o				
SLOPE OF LINE SEGMENT[1]	0.2	0.4	0.6	0.8	0.9
0.235	5.39	4.82	4.36	4.02	3.66
0.470	4.10	3.60	3.30	3.01	2.84
0.705	3.25	2.93	2.62	2.44	2.33
0.940	2.73	2.44	2.18	2.01	1.83
1.175	2.33	2.09	1.93	1.70	1.60
1.410	2.09	1.83	1.68	1.53	1.39
1.645	1.85	1.67	1.48	1.36	1.26
1.880	1.67	1.48	1.34	1.17	1.06
2.115	1.52	1.34	1.22	1.10	1.10
2.350	1.45	1.26	1.14	1.03	1.06
2.585	1.34	1.16	1.01	0.82	0.00
2.820	1.35	1.09	0.93	0.80	0.00
3.055	1.25	1.05	0.89	0.78	0.00
3.290	1.19	0.95	0.83	0.78	0.00
3.525	1.04	0.85	0.71	0.53	0.00
3.760	0.98	0.80	0.63	0.43	0.00
3.995	0.93	0.74	0.60	0.48	0.00
4.230	0.86	0.72	0.60	0.48	0.00
4.465	0.81	0.70	0.54	0.42	0.00
4.700	0.76	0.62	0.51	0.42	0.00

[1] Slope $\doteq - 6.22 \times 10^{-5} \times (TOC) - 2.03 \times 10^{-3} \times (ION) + 6.69$

TABLE 8. CADMIUM TRANSPORT RATES (cm/day) FOR FANNO c. FOR A SOLUTION SPEED OF 27.06 cm/day.

SLOPE OF LINE SEGMENT[1]	C/C_o				
	0.2	0.4	0.6	0.8	0.9
0.235	22.49	19.54	17.46	15.49	14.66
0.470	17.10	14.66	13.03	11.73	10.80
0.705	13.24	11.73	10.52	9.33	8.92
0.940	11.09	9.77	8.92	7.89	7.33
1.175	9.54	8.38	7.60	6.84	6.41
1.410	8.21	7.33	6.73	5.86	5.70
1.645	7.33	6.51	5.95	5.40	5.13
1.880	6.62	5.86	5.40	4.89	4.66
2.115	6.03	5.33	4.89	4.46	4.10
2.350	5.57	4.93	4.50	4.00	3.77
2.585	5.03	4.57	4.07	3.77	3.56
3.820	5.13	4.27	3.83	3.37	3.20
3.055	4.55	3.94	3.53	3.11	2.85
3.290	4.27	3.79	3.36	2.93	2.85
3.525	4.02	3.53	3.15	2.77	2.56
3.760	3.87	3.30	2.97	2.63	2.56
3.995	3.55	3.16	2.77	2.53	2.47
4.230	3.45	2.95	2.71	2.42	2.37
4.465	3.25	2.84	2.53	2.37	2.28
4.700	3.07	2.70	2.47	2.23	2.03

[1] Slope $\doteq -6.22 \times 10^{-5} \times (TOC) - 2.03 \times 10^{-3} \times (ION) + 6.69$

ANALYTICAL METHODS FOR LEACHATE ANALYSIS

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ABSTRACT

Several studies have included the various methods of analysis applied to sanitary landfill leachate. The first such report was a compilation of information available in the literature and from various active researchers in the field. The State of Illinois developed a similar document for monitoring landfill leachate in Illinois. The Canadian Government has developed a document presenting their analytical protocol also.

Although different analytical methods can be used to determine any specific parameter, the methods least subject to interferences should be chosen for leachates. Further research is necessary in analyzing variable strength leachate and leachates from different geographic locations.

Presently the Disposal Branch of the Solid and Hazardous Waste Research Division of the Municipal Environmental Research Laboratory (MERL) is sponsoring a "Round Robin" study on leachate analysis in order to get a better idea of precision and accuracy of selected methods. Additionally the Division has two on-going studies for evaluating methods applicable to hazardous waste analysis. These presently involve gross separation schemes with more refined analytical methodology to follow.

COLLECTION/STORAGE

The collection, transport, storage, and analysis of leachates is a very difficult task, if not downright frustrating.

Leachate can be collected from subsurface soil strata by using wells or piezometers placed in drilled holes. Leachate can also be collected above ground as it appears in springs or at the toe of a landfill, but commonly these samples contain eroded soil or may have reacted somewhat with the soil. Leachate may also reach the surface and enter surface waters via groundwater discharge. In this instance dilution plays a significant role in the chemical characteristics.

The gross chemical characteristics of leachate can be affected by the methods and materials used during sampling. Most researchers who have reported (1), (2), (3) indicate that a plastic container indicate that a plastic container is the collection vessel of choice. When collecting leachate in the field anaerobically, Streng (3) recommends the use of polyethlene containers. Further, Streng recommends polyethylene containers produced by a high pressure, non-catalyzed process. Chian and DeWalle (1) recommend polyethlene containers when analyzing heavy metal content collected leachate. Glass collection containers should be reserved for analysis of only those parameters where no absorption and/or adsorption can occur.

The storage of collected leachate is generally accomplished by refrigeration at 4°C in tightly stoppered containers. All researchers recommend certain analyses be made immediately as samples may be altered during storage under any conditions. Such analyses include those for oxidation-reduction potential (ORP), color, turbidity, suspended solids (SS), pH, and conductivity. Other parameters such as Chemical Oxygen Demand (COD) and organic nitrogen may also vary after sampling, but such changes can be reduced if the sample is acidified and stored at 4°C. Most researchers agree that storage times and conditions must be identified when reporting results.

ANALYTICAL PARAMETERS

The choice of parameters for leachate analyses should be based on the probability of detection of each parameter in the leachate (known as indicator parameters), as well as whether water quality standards exist for the parameter. To a certain extent, the type of waste at a given site also has a bearing on the selected parameters. Heavy metals and certain trace elements are normally included in the analysis of leachate because of their potential health hazards.

When a large number of samples must be analyzed, it is not feasible, or cost effective, to measure all parameters. As a rule of thumb those that can be done easily should be done initially. In their comprehensive study, Chian and DeWalle measured the following pollution parameters:

- o PHYSICAL PARAMETERS -- pH, ORP, conductivity, residue (total solids, TS; volatile solids, VS; dissolved solids, DS; and suspended solids, SS).
- o INORGANIC CHEMICAL PARAMETERS -- chloride, sulfate, phosphate, alkalinity and acidity, nitrate, nitrite, ammonia, sodium and potassium, calcium and magnesium, hardness, heavy metals.
- o ORGANIC CHEMICAL PARAMETERS -- COD, TOC, volatile acids, tannin and lignin, organic nitrogen.
- o BIOLOGICAL PARAMETERS -- BOD, coliforms.

Most information is obtained by measuring conductivity (which reflects salts and free volitile fatty acids), color or absorbance at 300 nm (which reflects iron and organics), and pH (a low value indicates presence of free volatile fatty acids). When more parameters are to be measured, they should include COD (which reflects concentration of organics) and TS (which reflects presence of organics and inorganics). After the above five parameters exceed a certain value, it is especially warranted to determine other parameters such as TOC, free volatile fatty acids, BOD, organic N, or specific anions and cations.

When an organic characteristic such as TOC or organic N is measured, the inorganic equivalent should also be included (the bicarbonate concentration and the ammonia concentration, for example). The ratio organic C/(organic C + inorganic C) then reflects the degree of biological stabilization of the sample, as acid fermentation followed by methane fermentation connects the complex organics to free volatile fatty acids, which are then converted into CH_4 and CO_2. The latter dissolves to a significant degree into the leachate and is reflected in the increased bicarbonate concentration. A high ratio would indicate little organic degradation, whereas a lower ratio would reflect increasing stabilization. A problem arises in that the only accurate method to measure the inorganic carbon is using a dual channel organic carbon analyzer or the inorganic channel. This is a piece of analytical equipment not found routinely in laboratories today.

METHODOLOGY SOURCES

For a complete listing of methods of analysis the paper by Chian and DeWalle (1) should be consulted. Generally, the methods are taken from Standard Methods (4) and Methods for Chemical Analysis of Water and Wastes (5).

CONCLUSIONS

Several pollution parameters of samples taken from a recently leached landfill vary immediately after being collected unless strict anaerobic sampling and storage conditions are maintained. Preliminary laboratory studies on physical, chemical, and biological parameters reveal that color, SS, and high salt content associated with leachates can interfere with colorimetric methods of chemical analysis. Interferences can possibly be circumvented by using a standard addition technique in which recovery is determined.

This obtained percent recovery is then used to readjust the measured value. Another avenue to remove interferences is the dilution technique. This is an attempt to dilute the leachate samples with increasing amounts of water to determine if the interfering effect can be reduced or overcome. Some researchers maintain this technique is less accurate than the standard addition technique.

It is beyond the scope of this paper to attempt to identify all potential problems involved in the analysis of leachate. The analyst must take all precautions identified in the method references, especially to be sure that interferences inherent to the sample and from external sources are recognized and eliminated or neutralized.

RESEARCH NEEDS/ACTIVITIES ONGOING

Based on the results of various other research efforts, the USEPA recently awarded a grant to Stanford University (7) to study the precision and accuracy for several methods of analysis of pollution parameters. It is expected that up to 5 pollution parameters will be extensively evaluated by some 25 laboratories. Statistical evaluation of the data generated will determine the precision of each of the methods, while comparison of different methods used to determine each parameter will further illustrate the accuracy of each test. The project duration is one year.

In a report entitled "Chemical Quality and Indicator Parameters for Monitoring Landfill Leachate in Illinois," Clark and co-workers(2) sampled 54 landfills in 35 Illinois counties. Thirty-seven parameters were analyzed in the leachate samples, and the results clearly showed the diverse nature of leachate quality. This study reaffirmed the recommendation of Chian and DeWalle that many samples of different strengths and from different locations should be investigated to substantiate the applicable method of analysis.

Further research is recommended to establish correlations between specific constituents and general constituents such as conductivity, absorbance at 400 nm, and pH. These three characteristics are easy to determine and can be valuable for monitoring and enforcement purposes; however, analysis for an extended list of pollutants is time consuming and expensive. Mooij(6) recommends a short, selected list of indicator parameters, chosen to represent the landfill, the surrounding environment, and the conditions under which leachate will travel. He further states that upon appearance of one or more of the selected parameters, more detailed analyses should be conducted. Chloride, hardness, iron, sulphate, and specific conductance have been used as indicator parameters. The recommendations of Mooij call for more detailed analyses parallel those of Chian and DeWalle.

Research is needed to establish the exact nature of interfering substances and potential method development to circumvent interferences. Using this approach Streng[3] found that interferences play a major role in chloride analysis involving any established methodology. Therefore, he set out to develop an indirect method for the determination of chloride, employing atomic absorption spectroscopy (AA). This indirect method requires no sample pretreatment and greatly reduces the time and effort required in analysis with no apparent loss of accuracy or precision. At this time the AA approach is the first reported effort of its kind and is still in the developmental stages. However, publication of the method is expected and interested researchers should contact Streng regarding this approach to chloride analysis in leachate.

In general, future research efforts should be directed to either developing methods of analysis that are not susceptible to the interferences found in leachates, or to finding ways of removing the interferences while using approve analytical methods. These removal techniques could include ion exchange, activated carbon treatment, coagulation effects, or lime dosing.

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LEACHATE TREATMENT BY BIOLOGICAL AND PHYSICAL-CHEMICAL METHODS -- SUMMARY OF LABORATORY EXPERIMENTS

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INTRODUCTION

In recent years, it has become well known that land disposal of solid waste in a humid climate will frequently produce leachate as a result of rainwater infiltrating the solid waste. This leachate often contains a high concentration of organic matter and inorganic ions. A number of incidents have been reported where leachate has contaminated the surrounding soil and polluted an underlaying groundwater aquifer or nearby surface water. One way to avoid or to correct such situations is to collect and treat the leachate.

The treatability of leachate has been reported by Boyle and Ham (1) using both aerobic and anaerobic biological processes. Aerobic and anaerobic stabilization of sanitary landfill leachate was also studied by Cook and Foree (7), and Rogers (19). A COD reduction between 58 and 99% was attainable by a single-step biological process. Boyle and Ham (1) and Foree and Reid (10) have also reported a two-step biological treatment process employing an anaerobic digester or filter followed by an aerated lagoon. Further reduction in COD, between 22 to 40 percent of the effluent COD of the first unit, has been accomplished by these investigators (1,10).

The use of chemical precipitation and oxidation methods for leachate treatment has been reported by Boyle and Ham (1), Karr (15), Cook and Foree (7), Ho, Boyle and Ham (13), Thornton and Blanc (22), Rogers (19), Simensen and Odegaard (21) and Roy F. Weston, Inc. (20). Other physical-chemical processes for leachate treatment, such as activated carbon and ion-exchange adsorption and reverse osmosis, have been studied by Cook and Foree (7), Ho, Boyle and Ham (13), Karr (15), Pohland and Kang (17), Van Fleet et al. (23) and Roy Weston, Inc. (20). The results of these physical-chemical methods of leachate treatment showed a wide variation of removal efficiencies which may well be due to the variable nature of the landfill leachates employed in these studies.

The purpose of this study was to summarize the results of all of the previous studies on leachate treatment and to devise a scheme of general applicability to treat leachate and to test this scheme systematically. The proposed scheme for leachate treatment under this study is given in Figure 1.

MATERIALS AND METHODS

The leachate employed in this study was collected from a lysimeter installed at the University of Illinois. This leachate contained a COD, TOC, and BOD of 49,300 mg/l, 17,060 mg/l and 24,700 mg/l, respectively, and had a pH of 5.63. The iron content of this leachate was as high as 2,200 mg/l.

The biological processes evaluated in this study included activated sludge, aerated lagoon and anaerobic filter. The

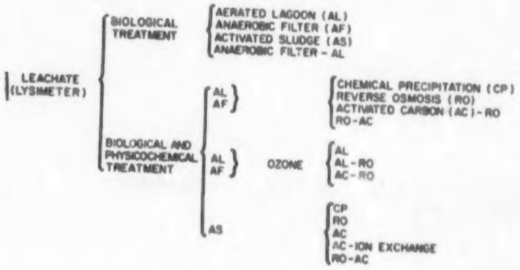

Figure 1. The Proposed Experimental Flow Diagram for Studies of the Promising Treatment Processes

activated sludge experiments were conducted with a test and a control unit operated in parallel and maintained at the same F/M ratio (g.BOD removed/g.MLVSS.day). The test unit, a 57-liter laboratory scale plug-flow activated sludge unit, received municipal sewage with a leachate addition of 0.5, 1, 2, 3 and 4 percent by volume while maintaining an F/M ratio of 0.3 day^{-1}. Two other F/M ratios, i.e., 0.6 and 1.0 day^{-1} were also studied while maintaining a leachate addition of 2 percent by volume. Comparisons of the sludge settling characteristics and the effluent BOD and COD were then made between the control unit and the test unit.

The aerated lagoon or the extended aeration process employed six completely mixed vessels fed with undiluted leachate with no sludge recycle. The tested hydraulic detention times, which were equal to the sludge ages, were 85.7, 60, 30, 14 and 7 days. These units were operated for periods from 70 to 150 days. The phosphorus requirements of the aerobic biomass were evaluated extensively, while phosphorus and nitrogen were added to the leachate in an amount of twice that needed for nutritional requirements.

A completely-mixed 56-liter anaerobic filter packed with "Surpac" slabs (Dow Chemical, Midland, MI) was used. The porosity of the filter was 94 percent with a specific surface area of 2.06 cm^2/cm^3. No nutrient addition was needed. The hydraulic detention studies were 7.5, 17.5, 42 and 74 days. Recycling of the effluent from the anaerobic filter was made to buffer the pH of the leachate fed to the unit and to keep the liquid inside the filter well mixed.

Physical-chemical methods were employed to treat both the raw leachate and the effluents from various biological processes. These processes were found to be not effective in treating the highly polluted leachate used in this study and further research was therefore concentrated on their applicability in the treatment of the effluent. Chemical precipitation experiments were conducted on effluents from the aerated lagoon and anaerobic filter using the standard jar test apparatus. Activated carbon studies were conducted with Filtrasorb 400 (Calgon, Pittsburgh, PA) with both batch dosages and columns. Ion-exchange adsorption studies were conducted with Duolite A-7 (Diamond Shamrock, Redwood City, CA), Amberlite IRA-938 and XE-297 HP (Rhom and Haas, Philadelphia, PA) in columns. Reverse osmosis studies on raw leachate and the effluent from the aerated lagoon were carried out with cellulose acetate and NS-100 membranes housed in a 3"-dia. stainless steel high-pressure test cell operated at 600 and 1500 psig under nitrogen atmosphere. A DuPont B-9 permeator (Wilmington, DE) operated at 400 psig was used to treat effluent from the anaerobic filter. Ozonation of biological effluent was carried out in a well-baffled one-liter glass reactor fed with O_3/O_2 mixture.

RESULTS AND DISCUSSION

Activated Sludge

The effect of increased leachate additions on influent BOD:P ratio and the effluent COD and BOD concentrations is summarized in Figure 2. The F/M ratio of these runs was maintained at 0.3 day^{-1}.

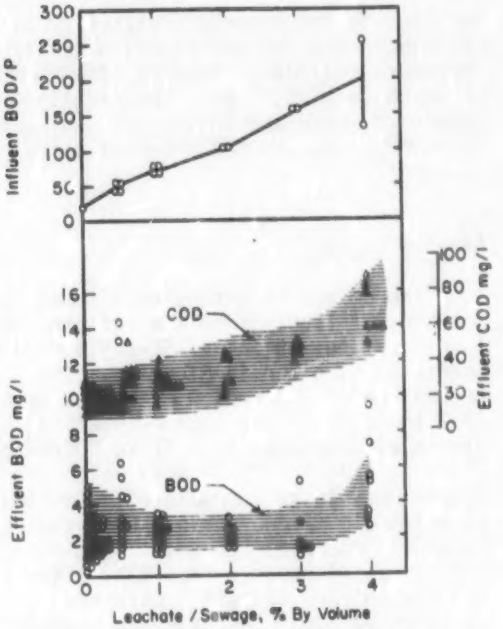

Figure 2. The Effect of Increased Leachate Additions on the Influent BOD/P Ratio and the Effluent COD and BOD Concentrations of the Test Unit

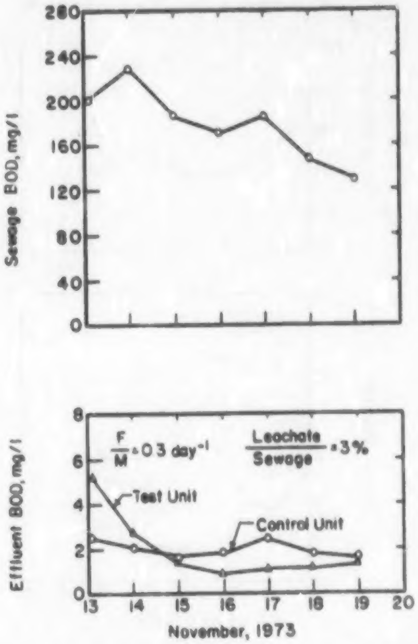

Figure 3. The Effect of 3% Leachate Addition on Effluent Quality of the Activated-Sludge Process

A comparison of the results of the test (Figure 2) and the control units (Figure 3) shows that increasing leachate additions does not greatly affect effluent BOD values although the COD values in the effluent are consistently higher from the test unit than that from the control. However, at 4% leachate addition, the BOD values in the effluent of the test unit increased with increasing time of operation. The test was discontinued at this leachate addition level because of deteriorating effluent quality. Boyle and Ham (1) found that leachate with COD of 10,000 mg/l could be added to domestic sewage in an extended aeration activated sludge unit at a level of at least 5% by volume without seriously impairing the effluent quality; however, the leachate in their study was 5 times weaker. The failure of the test unit at 4% by volume of leachate addition was attributable to a phosphorus deficiency as shown in Figure 2, where the BOD:P ratios in the influent varied from 130 to 255.

The effect of increasing loadings on the effluent characteristics was also studied at a constant leachate addition of 2%. The results showed that the BOD of the test unit was consistently higher than that of the control unit at an F/M ratio of 0.6 and 1.0 day^{-1}. At the highest loading of 1.0 day^{-1} studied, the effluent BOD of the test unit (Figure 4) increased noticeably with time, indicating that the activated sludge unit is preferably operated at low F/M ratios when receiving leachate additions.

The sludge settling characteristics show that the addition of 0.5% leachate resulted in a decreased settling rate of the sludge interface; increasing the leachate addition (up to 3%) at the same loading (i.e., 0.3 day^{-1}), however, did not further deteriorate the settling characteristics (Figure 5). The impairment of the sludge settling was even more noticeable at the higher sludge loading, e.g., 0.6 day^{-1}. The presence of high concentration of iron in leachate with resulting low soluble phosphate concentration may have contributed to the impaired sludge settling upon leachate addition. Therefore, the addition of even a small amount of leachate to the activated sludge process would tend to impair the sludge

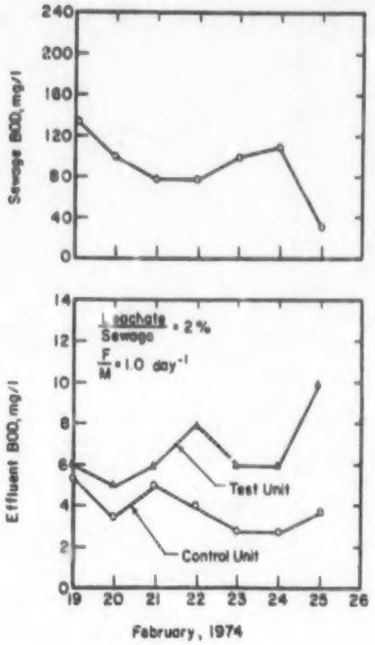

Figure 4. Effluent Quality During 2% Leachate Addition at the 1.0 F/M Ratio

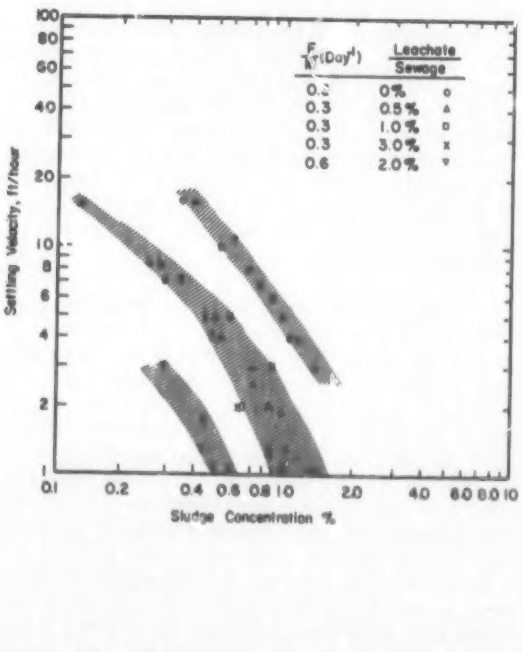

Figure 5. The Effect of Leachate Addition on the Sludge Settling Characteristics

settling in the secondary clarifier of an existing plant and additional capacity is therefore required. However, the advantage of leachate additions is the resulting low levels of phosphate in the effluent of the test unit, i.e., on the order of a fraction of 1 mg/l.

Aerated Lagoon

The effect of detention time and thus the organic loadings on the effluent TOC concentrations during a 150-day period of operation is given in Figure 6. The percentage of TOC reduction varies from 98.2 to 99.2% as the organic loading decreased from 1.94 to 0.67 kg TOC/M^3 day (5.61 to 1.94 kg COD/M^3 day) which corresponds to an increase of detention time from 30 to 85.7 days. Cook and Foree (7) reported COD stabilization efficiency of greater than 97%. Karr (15) and Pohland and Kang (17) obtained variable effluent qualities while operating at lower detention times ranging from 2

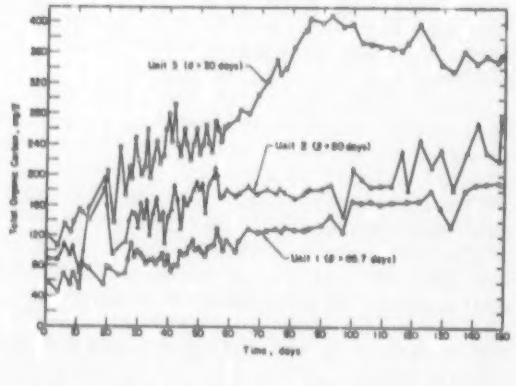

Figure 6. Total Organic Carbon in Effluent of Aerated Lagoons 1, 2 and 3 Treating Leachate

to 15 hours and 2.3 to 8 hours, respectively, and organic loadings varying from 4.5 to 54.2 kg COD/M^3 day and 1.5 to 5.3 kg COD/M^3 day. The mixed liquor volatile suspended solids (MLVSS) decreased from 12,000 to 8,000 mg/l as the detention time increased from 30 to 85.7 days. The high suspended solid content in the lagoon requires intensive mixing and aeration of these units. The kinetics constants determined from these runs are 0.42 mgVSS/mgCOD, 0.025 day^{-1} and 4.9 x 10^{-4} (mg/l VSS)$^{-1}$ (day)$^{-1}$, respectively, for the yield, the microorganism-decay coefficient and the overall first-order substrate removal rate constant per unit weight of microorganism which agreed well with that reported by Cook and Foree (7).

An extensive evaluation of phosphate requirements showed that the COD:P ratio in the influent of the 30-day unit should be at least 300:1. For units with a retention time of 85.7 and 60 days, they were even able to be operated with a COD:P ratio of 1540:1 in the feed solution. Cessation of nutrient addition at a COD:P ratio of 165:1 to the units operated at relatively low detention times, e.g. 7.5 and 15 days, caused an immediate increase in effluent organic matter, a decrease in biological MLVSS and a deterioration of sludge settling rates.

All units operated at various detention time showed high removals of heavy metals, especially for iron (99.9%), zinc (99.9%), calcium (99.3%) and magnesium (75.9%). Lower removals were observed for sodium (24.1%) and potassium (17.0%). The high removals of heavy metals appear to be related to chemical precipitation and flocculation in the aerated lagoon whereas the low removals for sodium and potassium appear to be biologically mediated.

Since an appreciable amount of sludge was produced from the aerated lagoon, considerations were also given to sludge disposal. The dewatering characteristics of the sludge from the 30-day unit were greatly improved by the addition of cationic polymers (Primafloc C7, Nalco 73C32) and inorganic coagulants. An approximate 20 times decreases in the specific resistance of the sludge was obtained at polymer dosages varying between 0.15% to 1.5% and inorganic dosages of 2.9% to 25.5% on the basis of dry sludge weight. These decreases in the specific resistance corresponded to a 6-time increase in vacuum filter yields.

Anaerobic Filter

Studies on the effectiveness of the anaerobic filter in treating leachate were conducted for a period totaling 518 days. During the initial phase of the study, which lasted 218 days, the different start-up procedures, pH stabilities and shock loadings of the unit were tested. In phase II of the study, which lasted 250 days, the various operational difficulties of the completely mixed unit were evaluated. In the last phase of the anaerobic filter experiment, which lasted 50 days, the effect of organic loadings on the effluent characteristics was studied.

The recirculation ratio of the anaerobic filter effluent was studied after one volume turn over, i.e., 42 days of operation. Based on the acid (sulfuric acid) titration of the effluent, five parts of effluent were required to increase the pH of the influent to 7.0. Therefore a minimum ratio of 1:6, i.e., one part of leachate sample to five parts of effluent, was required. In order to operate the unit safely and well-mixed the ratio of 1:20 was maintained, resulting in a complete turnover of the liquid volume in the unit every 1.8 days. Since the time required for mixing was short as compared to the hydraulic detention time, i.e., 1.8 days versus 42 days, the unit can be considered completely mixed. The use of a completely-mixed anaerobic filter will allow utilization of the entire length of the filter column for substrate removal; in plug flow units, however, only the lower one meter (3.3 ft.) is generally effective in substrate removal according to Young and McCarthy (24,25) and Jennett and Dennis (14). When the unit was loaded with 6, 10 and 14 times the initial rate of leachate addition, the initial decrease in pH and upsurge in gas production leveled off after about six days of operation, indicating good stability of the unit under conditions of various shock loadings.

During the initial stage of the phase II study, 97% of the COD was removed at a loading of 0.62 kg COD/M^3 day. Up to 89% of COD was removed in the form of methane gas leaving the system, whereas the remainder of COD was present in the form of inorganic carbon in the solution. Only a small amount of COD was removed as biomass accumulated in the anaerobic filter. The low yield of biomass, i.e., approximately 0.012 g biomass produced per g of COD removed, allowed the filter to be operated without nutrient addition. Due to the low yield of biomass, the COD:P and COD:N ratios in the feed could be maintained at 4360:1 and 39:1, respectively, without impairment of the filter operation. Any P requirement was sufficiently met by the addition of the anaerobic digester sludge added to the unit initially for seeding.

Heavy metal toxicity was observed which resulted in the COD removal decreased to as low as 64% with the unit during phase II of the study. Analysis of the effluent showed a gradual increase in soluble heavy metals from an undetectable amount to as high as 2.8, 0.9, and 0.2 mg/l respectively for Fe, Cu and Zn. High Cu content may have contributed to the malfunction of the anaerobic filter. This was, however, eliminated after addition of 75 mg/l Na_2S to the column content. The sulfide addition caused a decrease in oxidation reduction potential (ORP) values while reducing the heavy metal concentrations.

In view of the lower installation and operation costs of the anaerobic filter as compared with those of the aerated lagoon (16), it is obvious that the anaerobic filter is a preferred biological process of treating high strength leachate. In addition, the absence of any nutrient additions, the extremely low yield of biological solids and the production of useful energy, i.e., methane gas, strongly supports the use of the anaerobic filter for leachate treatment. As a result of this study, a pilot-scale 25,000 gal. anaerobic filter was constructed by the City of Enfield, CT, funded by OSWAMP (US-EPA), and is currently being evaluated for treating leachate generating from the landfill site located in the Township.

Physical-Chemical Treatment

Since the raw leachate generated from a recently installed landfill consists mainly of low molecular weight volatile fatty acids (6), it is more amenable to the various biological treatment processes than to physical-chemical processes. Neither activated carbon nor ion-exchange resins will adsorb these low molecular weight volatile fatty acid effectively (11). Removal of volatile fatty acids is also quite low with the reverse osmosis process (8) without pH adjustment. Although adjustment of pH to higher values, e.g., 9, would increase the membrane rejection of these organic compounds (9) appreciably, the high buffering capacity of the raw leachate renders this economically unattractive. Ozone oxidation of these volatile fatty acids was also found to be slow under the prevalent acidic pH of the leachate (4,12). The use of an excessive amount of chemical coagulants, e.g., up to 2,700 mg/l, to chemically precipitate the raw leachate resulted in only less than 30% removal of COD (7,13,15,21,22,23). However, after treating the leachate with biological processes, the residual organics in the treated effluent consist mainly of high molecular weight carbohydrate, humic and fulvic acid complexes (2,5) and are therefore more amenable to removal by physical-chemcial processes. These effluent organics are characterized by lower BOD:COD and COD:TOC ratios as compared with the raw leachate due to removal of organics with a high BOD/COD and COD/TOC ratio in the biological process.

The above conclusion has been borne out by several studies on the treatability of leachate. While Boyle and Ham (1) noted a 93% COD removal in anaerobic digesters, Ho, Boyle and Ham (13), using a similar leachate, obtained only a 34% COD removal rate using activated carbon at a maximum dosage of 20,000 mg/l; or a 59% removal of COD using a granular activated carbon (Filtrasorb 400) column having a detention time greater than 20 minutes. The leachate studied, having a high BOD/COD and COD/TOC ratio, was generated from a relatively recent fill as indicated by its high free volatile fatty acid content, i.e., 65% of the COD organics. After biological treatment of a similar leachate, Pohland and Kang (17) obtained a COD removal in the effluent of as high as 91% using activated carbon, indicating that biological treatment followed by carbon adsorption will result in high organic removal. Cook and Foree (7) also obtained a high percentage of COD removal, i.e., 70%, while treating aerated lagoon effluent with activated carbon. Roy F. Weston, Inc. (20) on the other hand, observed an 89% COD removal with leachate collected from a stabilized landfill site indicating that when the landfill has been subject to extensive biological stabilization, it is not necessary to treat the leachate by biological treatment processes prior to the activated carbon treatment. Since stabilized leachate consists mostly of high molecular weight humic substances as characterized by a low BOD/COD and COD/TOC ratio, it is to be expected that activated carbon will remove these compounds effectively. Results of the present study show that 70% of COD in effluent of the aerated lagoon and aerated effluent of anaerobic filter can be removed by the carbon columns. Up to 50% of the COD in the effluent of anaerobic filter were removed indicating that leachate was stabilized to a lesser extent with the anaerobic filter (42-day hydraulic detention time) as compared with the aerated lagoon (30-day hydraulic detention time). Only 50-58% removal of organics in aerated lagoon effluents was accomplished by ion-exchange resin (3,17) indicating its inferiority to activated carbon processes.

It was found that reverse osmosis like activated carbon also removes the high molecular weight carbohydrate humic complexes more effectively than the free volatile fatty acids and is thus more suitable for treating biologically stabilized leachate effluent and leachate obtained from stabilized landfills. Whereas the present study noted only a 56% removal of organics with raw leachate generated in a recent fill, as much as 95% COD removal was obtained in anaerobic filter effluent using the conventional cellulose acetate membrane (at a product water recovery of 50%). In both cases better removal of organics with reverse osmosis can be obtained using membranes having better capabilities of rejecting organics, such as the newly fabricated DuPont's B-9 permeator. Up to 98% removal of organics was obtained with the aromatic polyamide membrane used in the B-9 permeator at a greater percentage of product water recovery, i.e., 75%. Using a leachate collected from a stabilized landfill site, Roy F. Weston, Inc. (20) was able to accomplish 80% removal of organics at a product water recovery rate of 80%.

Chemical oxidation of raw leachate as well as effluent from biological units employing calcium hypochlorite and ozone showed that relatively little COD could be removed even with 3 to 4 hours of oxidation (3,13,15). For leachate collected from a nearby stabilized landfill site, Roy F. Weston, Inc. (20) observed no removal of organics with calcium hypochlorite whereas 22% removal of COD was obtained with ozonation after 4 hours. Up to 48% and 37% removal of COD was observed respectively in the present study with ozonation of aerated lagoon and anaerobic effluent. Due to the low COD removal as well as the high costs of these chemical oxidants, chemical oxidation does not appear to be feasible for leachate treatment.

Of all the physical-chemical methods evaluated by the previous investigatiors and in the present study, reverse osmosis membrane treatment was found to be more effective in removal of organics, especially when applied to biologically treated and highly stabilized leachates. Special precautions, however, may have to be used to prevent membrane fouling. The membrane process is followed in effectiveness by activated carbon treatment of biologically treated and stabilized leachate.

Since treatability of leachate is closely related to the chemical composition of leachate, the organic matter removal with each process can be related to COD/TOC and BOD/COD ratios. Figures 7 & 8 summarize treatment data from this study as well as those reported in the literature. It should be noted that the envelopes in Figure 7 and 8 for the percentage of COD removal versus BOD/COD and COD/TOC ratios only serve to indicate the general trends of COD removal that would occur with different BOD/COD and COD/TOC ratios in raw leachate or biologically stabilized leachate and do not have any statistical significance. The individual envelope was generated using extreme values for its boundary. They are, therefore, useful to

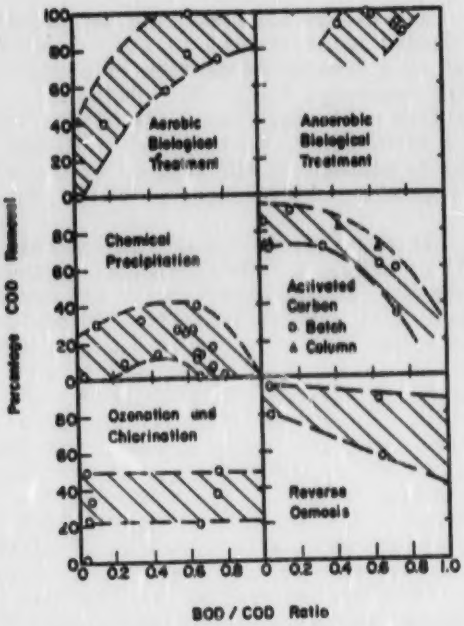

Figure 7. Percentage of COD Removal by Various Treatment Processes Versus BOD/COD Ratio

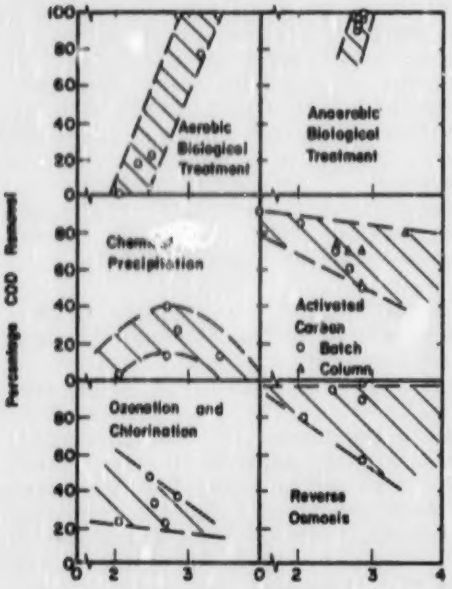

Figure 8. Percentage of COD Removal by Various Treatment Processes Versus COD/TOC Ratio

select treatment processes that are optimum for treating a specific leachate with known ratios of BOD/COD or COD/TOC. Follow-up laboratory experiments, however, are recommended to ascertain the selection of each of the specific treatment processes.

CONCLUSIONS

High strength leachate (e.g., COD > 5,000 mg/l) generated from recently installed landfills is more amenable to biological treatment processes. This leachate is characterized by high ratios of both BOD/COD (> 0.4) and COD/TOC (> 2.8). Low strength leachate (e.g., COD < 1,000 mg/l) generated from a stabilized landfill or present in effluent of biological units treating leachate is more amenable to physical-chemical treatment processes. This leachate is generally characterized by a low ratio of either BOD/COD (< 0.2) and/or COD/TOC (< 2.5). Of all biological processes evaluated, the anaerobic filter was found to be the most economical process due to its lower capital and operating costs. Reverse osmosis and activated carbon were found applicable to remove residual organic contaminants in effluent of biological leachate treatment units.

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LEACHATE TREATMENT BY SOIL METHODS

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Abstract

This paper examines the mechanisms of contaminant attenuation active in experiments on soil - landfill leachate contact. Ten soils were used in both column and dispersed soil reactor (DSR) configurations. Anoxic conditions were maintained. Leachate was allowed to pass through the soils and contaminant reduction during passage was measured.

Sorption processes including ion exchange were the most active of the mechanisms considered. They influenced the removal of most cations present as well as organic matter (COD). Removal isotherms for NH_4, Fe^{2+}, Zn^{2+}, K^+ and COD showed positive correlation with cation exchange capacity of the soil. The release of Ca^{2+} and Mg^{2+} during contact identified the existance of ion exchange.

Dispersion and dilution actively influenced contaminant concentration. Asymmetric breakthrough curves suggested incomplete contaminant mixing in non-migrating water.

The concentrations of Zn^{2+}, Fe^{2+} and Mn^{2+} in the leachate exceeded predicted solubilities as pH increased to neutrality during soil contact. The potential for removal by precipitation appeared to exist.

The destruction of organic matter through anaerobic microbial action was limited by incomplete culture development.

Filtration functioned to remove small amounts of suspended solids in the leachate and any precipitates formed. Gaseous exchange was assumed to exist as the partial pressure of carbon dioxide reduced to atmospheric conditions.

The stability of these mechanisms was tested by the passage of water through the soils after leachate contact. Little or no release was shown for Fe^{2+}, Mn^{2+} and Zn^{2+}, slight to moderate for NH_4^+ and K^+ and substantial for COD and Na^+.

The use of DSR data in modelling contaminant removal during continuous flow was demonstrated.

INTRODUCTION

Since the early 1970's, there has been a dramatic increase in research dealing with groundwater contamination from solid and liquid waste disposal in soil. On-site investigations have identified the complete spectrum of environmental impacts ranging from disposal with little or no deleterious effect to the production of highly toxic conditions within the groundwater system. In the majority of cases involving the landfilling of municipal solid wastes, the impact upon the groundwater system has been tolerable. This has been generally attributed to the attenuation of contaminant concentrations during leachate movement through the soil.

Of the mechanisms associated with contaminant attenuation, the most frequently cited are (1, 2, 3);

1. dispersion and dilution
2. sorption including ion exchange
3. precipitation
4. biological transformation
5. gaseous exchange
6. filtration.

The properties of the soil environment that influence the extent to which these mechanisms are operative appear to include (1, 2, 3):

1. soil grain size
2. organic content
3. cation exchange capacity
4. pH
5. Eh
6. hydrous oxides
7. free lime content

The relative importance of one property over another is not well documented. It is likely to vary from one situation to the next.

A programme of research into landfill contaminant attenuation in soil has been underway at the University of Waterloo for several years. Its purpose has been, in part, to identify the types of mechanisms involved and the extent of this involvement. The paper presented here describes some of the results from this research.

EXPERIMENTATION*

Bench-scale experiments were performed in the laboratory for the study of soil-contaminant interactions. Ten soils were collected at depths from 1m to 8m at several landfill sites in Southern Ontario. The ranges of properties exhibited by these soils are summarized in Figure 1. A positive correlation existed between cation exchange capacity (CEC) and soil clay content as estimated by the % dry weight of soil less than 0.002mm particle size. The influence of increased CEC with increased organic matter in the soil is also shown. Soil pH values fell between 7.1 and 7.4 inclusive. The clay minerals illite and chlorite were present in all soils; quartzite in all but soils 5, 6 and 10; kaolinite in Soils 1, 2, 3, 4 and 6; montmorillonite and vermiculite in soils 2, 3 and 4. The dominant resident ion on all soils was Ca^{2+} followed by Na^+, Mg^{2+} and K^+ in that order.

The leachate used for contact with soil was generated in a lysimeter charged with residential solid waste (4). Typical concentrations for the period of study are shown in Table 1.

Leachate-soil contact was achieved in packed columns (7.3cm ID by 40cm in length) and dispersed soil reactors (DSR; 5.2cm ID by 25cm in length). Leachate was added to the top of the column and allowed to percolate down through the soil. Effluent samples were collected and analysed for their chemical composition. For the DSR's, soil and leachate were added, shaken, settled and separated with the residual liquid analysed as above. The liquid was then passed on to the next in the series of 10 reactors and the process repeated. This was done to preserve the influence of one contaminant on another with respect to interactions with soil. All activities were performed under N_2 gas to maintain anoxic conditions.

Estimates of contaminant attenuation were made and analysed in terms of operative mechanisms. A discussion of these mechanisms is presented below.

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*The experimental methodology employed in this research has been presented in detail elsewhere (3).

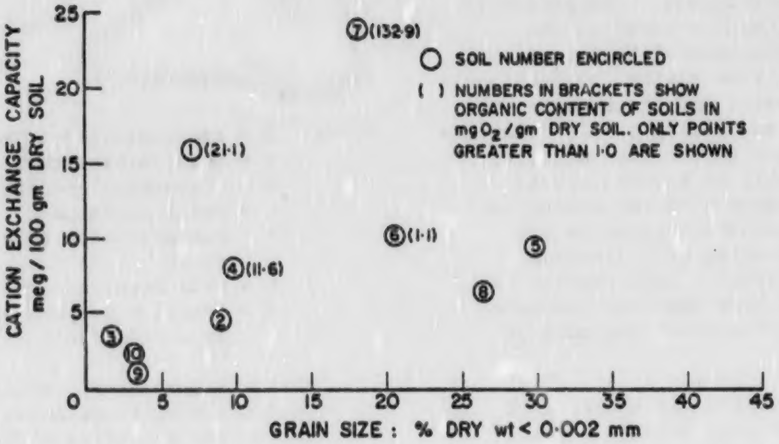

FIGURE 1. COMPARISON OF SOIL PROPERTIES

PARAMETER	CONCENTRATION (mg/ℓ)	PARAMETER	CONCENTRATION (mg/ℓ)
TOTAL DISSOLVED SOLIDS	16,024	Ca^{2+}	630
VOLATILE DISSOLVED SOLIDS	9,586	Mg^{2+}	118
COD	31,490	Na^{+}	511
BOD_5	18,470	K^{+}	559
NH_4^{+}	915		
ORGANIC - N	108	Fe^{2+} [1]	71
pH	5·65	Mn^{2+}	27
TANNINS & LIGNINS	200	Zn^{2+}	21
		Cu^{2+}	0·26
Cl^{-}	819	Pb^{2+}	1·3
SO_4^{2-}	790	Ni^{2+}	0·61
NO_3^{-} (as N)	0·25	Cr (IV)	0·07
TOTAL DISSOLVED P	23		

[1] METALLIC IONS WERE ASSUMED TO EXIST IN THE REDUCED FORM

TABLE 1. TYPICAL LEACHATE COMPOSITION

Dispersion and Dilution

Chemical concentrations in groundwater are reduced by combinations of dispersion and dilution. Dispersion is generally explained in terms of the "bundles of pore channels" concept where, in the laminar flow regime, pore channels of varying lengths and sizes cause dissemination and mixing of flow elements (5). As a front of leachate migrates through the soil, it becomes spread through dispersion with the production of the characteristic concentration gradients decreasing with distance away from the front. This results in concentrations less than the leachate concentration throughout the zone of dispersion.

For the addition to the soil of small pulses of contaminated liquid, dispersion can provide effective concentration reduction. For step function additions such as the movemment of leachate from a landfill, the effectiveness of dispersion is lessened as the length of the step function increases. This is due to the eventual attainment of full leachate strength. The effects of dispersion are often observed at landfill sites (6, 7, 8).

Dilution of groundwater contaminants can exist through the presence of slow moving and or stationary pore water in films or in "dead-end" pores. Depending on flow conditions, contaminants driven by concentration gradients diffuse into this stationary water thus retarding the movement of the contaminant. This is the same principle that forms the basis for thin-layer chromatography and is referred to as the "chromatographic effect" in subsequent discussions. As the rate of flow past this stationary liquid increases, the extent of molecular diffusion reduces with the arrival of the moving front at a point of observation well before expected. Under slow flow conditions with molecular diffusion maximized, the moving front will reach an observation point at a time suggesting that the whole pore volume had participated in the flow. This has been shown with a one dimensional convection-dispersion - capacitance model by Coats and Smith (5). The model in differential form for a conservative tracer is;

$$D \frac{\partial^2 C}{\partial n^2} - v \frac{\partial C}{\partial n} = f \frac{\partial C}{\partial t} + (1-f) \frac{\partial C^*}{\partial t}$$

and

$$(1-f) \frac{\partial C^*}{\partial t} = k(C-C^*)$$

where: C = concentration in mobile fluid
v = mean interstitial velocity
D = dispersion coefficient
n = axial distance
C* = concentration in stationary fluid
k = rate constant
f = fraction of pore space occupied by mobile fluid

This chromatographic effect functions to retard the moving contaminant front and is particularly effective with fissure flow in fractured bedrock (9). The extent to which it is operative in any specific case is difficult to determine since the fully-developed chromatographic effect gives results similar to those which would be obtained if the complete pore volume were participating in the flow.

If the diffusion of contaminants into the stationary volume is incomplete, the face of the moving contaminant front becomes assymetric being skewed in a direction toward the front. This represents a further reduction in contaminant concentration across the contaminant front.

Molecular diffusion may also be operative in spreading contaminants particularly in directions normal to fluid movement. Molecular diffusion is a slow process and its effect would be significant primarily in cases with small contaminant pulses and sufficient time for diffusion to occur.

Results from column experiments with Soil 2 are shown in Figure 2. They exist in the form of contaminant breakthrough curves (BC). The chloride ion (Cl^-) was taken to be a fully conservative tracer in this research. Support for this assumption is provided subsequently. The BC for Cl^- was spread and as such showed the influence of dispersion in reducing contaminant concentrations at the leachate front.

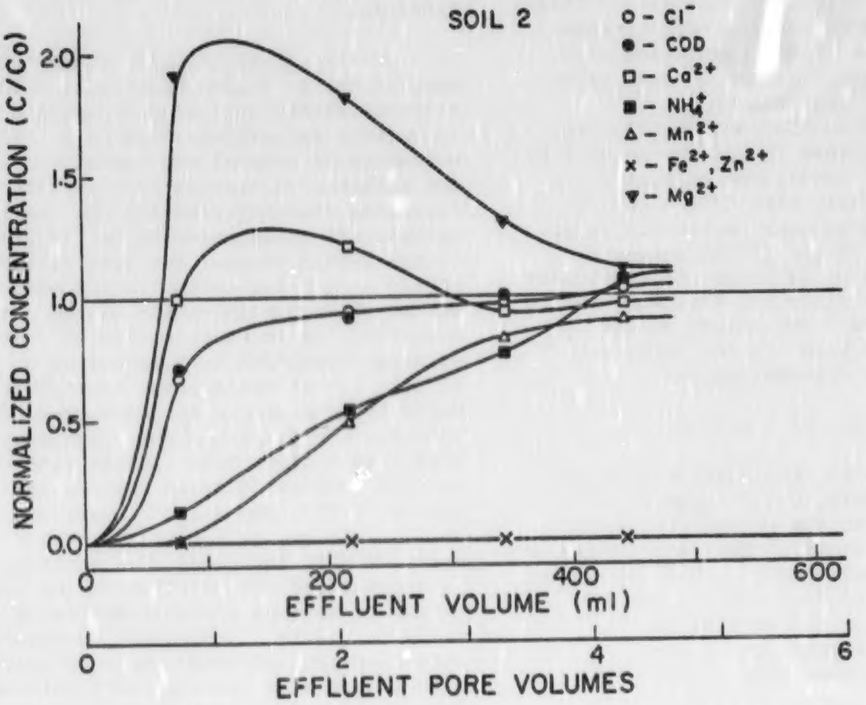

FIGURE 2. BREAKTHROUGH CURVES FOR SOIL 2, COL 1

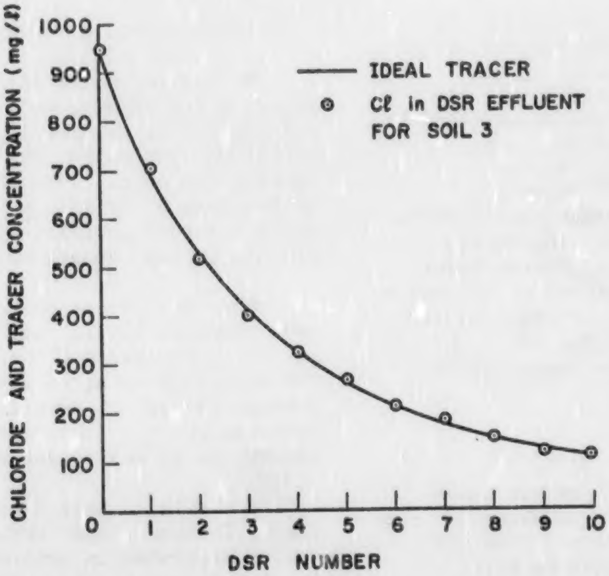

FIGURE 3. COMPARISON BETWEEN CHLORIDE ION AND IDEAL TRACER CONCENTRATION

The properties of the Soil 2 column included in part, pore volume (PV) = 104cc, 88% saturation before leachate passage and approximately 100% saturation after leachate passage. It can be seen from the BC for Cl^- that the C/Co = 0.5 (normalized concentration) occurred at 0.64 PV well before the discharge of 1 PV. This was under conditions of near saturation. Thus, with C/Co = 0.5 at 0.64 PV some stationary water did exist. The assymetry of the Cl^- BC suggests that diffusion of Cl^- into the stationary water did take place but that equilibrium was not reached. The actual volume of the stationary water is not known but from the above discussions was

$$\leq (1 - 0.64) \text{ PV} = 0.36 \text{ PV}$$

In a mass balance performed on the breakthrough data, 0.125 mg/gm of Cl^- were retained in the column during the passage of 2 pulses of leachate (Figure 2 deals with 1 pulse only). Had the entire pore volume including the stationary water been saturated with Cl^- at the leachate concentration, the mass retained would have been 0.155 mg/gm. This suggests on an equivalent basis that .81 PV were saturated with Cl^- at the leachate concentration and, thus, molecular diffusion of Cl^- into the stationary water was not complete.

Subsequent studies to desorb the columns with water released 0.119 mg/gm of Cl^- as compared to 0.125 mg/gm. originally retained. This favourable comparison attested to the conservative property of the chloride ion. The results of additional Cl^- tracer experiments are shown in Figure 3. Leachate was passed through the 10 DSR's in series. The curve represents the concentration of an ideal tracer based on the initial concentration of Cl^- and a knowledge of the soil moisture contents and their Cl^- content. The excellent fit further supports the conservative nature of the Cl^- ion.

Sorption

In this work, sorption was taken to include all processes in which contaminants are transferred from the liquid phase onto or into the solid phase. Most of the research to date on the sorption of leachate contaminants by soil has dealt primarily with cation exchange.

Griffin et al. (1, 10) found that many of the so-called heavy metals were actively sorbed onto clay minerals mixed in various proportions with sand. The mechanism of removal was identified as ion exchange increasing in intensity with increased concentration and pH. Cation removal was accompanied by Ca^{2+} release to the liquid phase. For some of the metals a limiting concentration was defined beyond which precipitation of the cation occurred. As an example, for Pb^+ ion exchange prevailed as a mechanism up to pH near 7.5 at which point precipitation began to dominate as the mechanism of removal. Sorption isotherms for many metals were presented. Metals removed in this fashion included Pb, Cd, Zn, C and Cr (III). The anionic heavy metals, Cr (IV), As and Se exhibited reduced sorption with increased pH. Korte et al. (11) described the sorption of ten heavy metals on various classifications of surficial soils. Subsequent to contact with comparatively weak leachate spiked with single heavy metals, soil columns were segmented and extracted with water and with 0.1 N HCl. Generally less than 3% of the sorbed metal was extracted with water. The ease of extraction with water was ranked as:

V >Se>As>Cr>Zn>Ni>Cd>Hg>Cu>Pb.

Fuller and Korte (2) reported that pH, free ion oxides, soil particle size and fluid flux had significant effects on cation removal while CEC and soil organic content had none. This is in contrast to the work of Griffin et al. (10) in which a strong positive correlation between CEC and cation removal was demonstrated.

Sorption processes beyond ion exchange have received little attention. Griffin et al. (10) observed that very little COD reduction occurred during leachate passage through their clay-sand mixtures. Davidson et al. (12) reported Freundlichian adsorption of pesticides on soil.

The BC's of Soil 2 presented in Figure 2 suggest that sorption was an active mechanism in contaminant attenuation in this research. Very similar BC patterns were exhibited by four additional soils for which column experiments were completed. The concentrations of both Ca^{2+} and Mg^{2+} in excess of the influent leachate concentrations signaled the release of these two cations from the soil and identified the existence of cation exchange.

Contaminant attenuation by all processes operative within the soil columns was calculated by comparing the BC for the ion in question with the Cl^- BC. By this method, the amount of Ca^{2+} and Mg^{2+} desorbed in Soil 2 for the length of the BC's given in Figure 1 was calculated to be 6.94 meq. The CEC of Soil 2 was 4.5 meq/100 gm or 29.4 meq/column.

The data in Table 2 represent cation removal for Soil 2. Since more cations were removed on an equivalent basis than released, the existance of addition attenuating processes was apparent. The cations NH_4^+, Mn^{2+}, K^+ and Na^+ (K^+ and Na^+ are not shown in Figure 2) reached their initial leachate concentration within the period of investigation. This indicated that the process(es) responsible for their removal was regulated by a maximum capacity. Sorption would represent such a process. Of the 14.6 meq of cations removed, 6.9 meq appear to have been removed by ion exchange. The balance were removed by some other sorption process(es).

TABLE 2. CATION REMOVAL IN SOIL 2

CATION	LEACHATE CONC. mg/l	CATION REMOVAL (AFTER 4.6 PV) meq/column
NH_4^+	846	8.2
K^+	532	2.2
Na^+	732	2.7
Fe^{2+}	91.9	1.2
Mn^{2+}	28.4	0.2
Zn^{2+}	15.3	0.1
Ca^{2+}	300	− 1.2
Mg^{2+}	168	− 5.7
pH	5.75	7.05

Figure 2 shows that no Fe^{2+} or Zn^{2+} appeared in the effluent from the columns. Thus it may have been that precipitation was the mechanism of removal for these two metals.

Some removal of organic matter within Soil 2 did take place despite the fact that Cl^- and COD BC's in Figure 2 are almost coincident. This was due to the very high initial COD = 28,788 mg/l. It will be shown subsequently that little evidence of microbial activity within the column could be demonstrated. However, limited sorption of organic matter did occur. The mass of COD removed was 101 mg COD/100 gm soil (661 mg/column) in contrast to 125, 52, 34, 22 and 3 mg/100 gm soil for NH_4^+, Na^+, K^+, Fe^{2+} and Mn^{2+}.

Breakthrough curves for soils 1, 3 and 10 are shown in Figures 4, 5 and 6. Many of the characteristics identified for Soil 2 are visible in the curves for these soils:

1. desorption of Ca^{2+} and Mg^{2+}
2. limited removal of COD
3. moderate removal of NH_4^+ and Mn^{2+}
4. extensive removal of Zn^{2+} and Fe^{2+}

It is important to note that partial breakthrough of Fe^{2+} occurred in Soil 3 and complete breakthrough in Soil 10. Partial breakthrough of Zn^{2+} also occurred in Soil 10. This suggests that, for these 2 soils in any case, the removal of Fe^{2+} and Zn^{2+} was taking place by means of some exhaustible process such as sorption.

Comparisons of contaminant removals are shown in Table 3. For COD, Fe^{2+}, K^+, Mn^{2+} and NH_4^+, there appears to be a general trend of increased removal with increased CEC although discontinuities do exist. The determination of correlation coefficients was negated by the scarcity of data.

From the foregoing, it would appear that sorptive processes including ion exchange played an important, and in many cases, dominant role in contaminant removal.

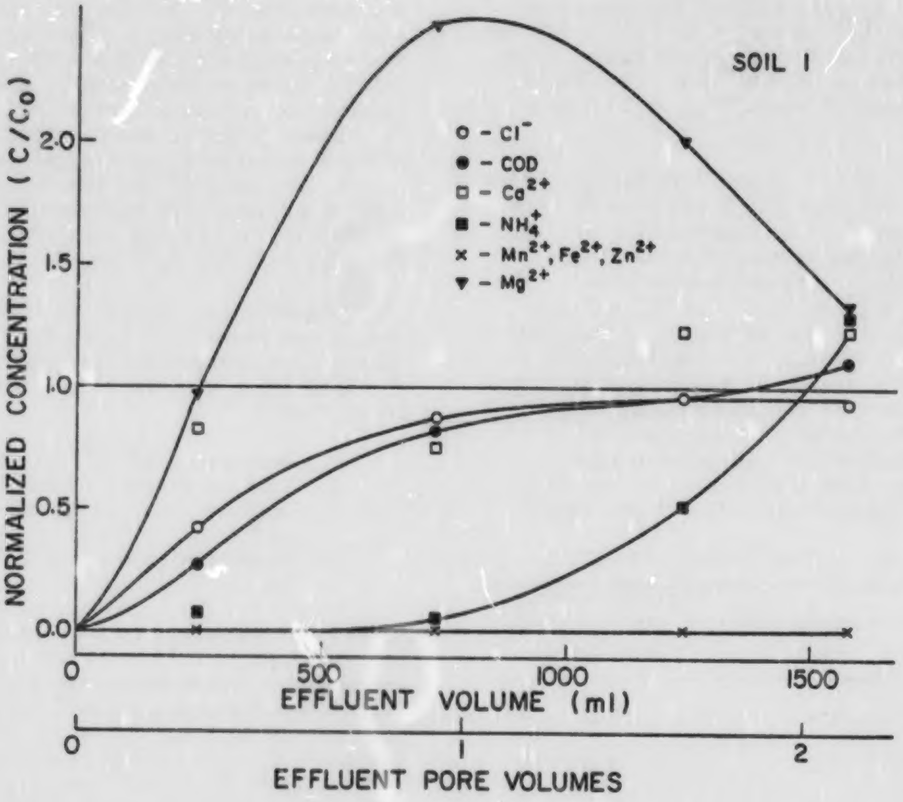

FIGURE 4. BREAKTHROUGH CURVES FOR SOIL 1, COL 1

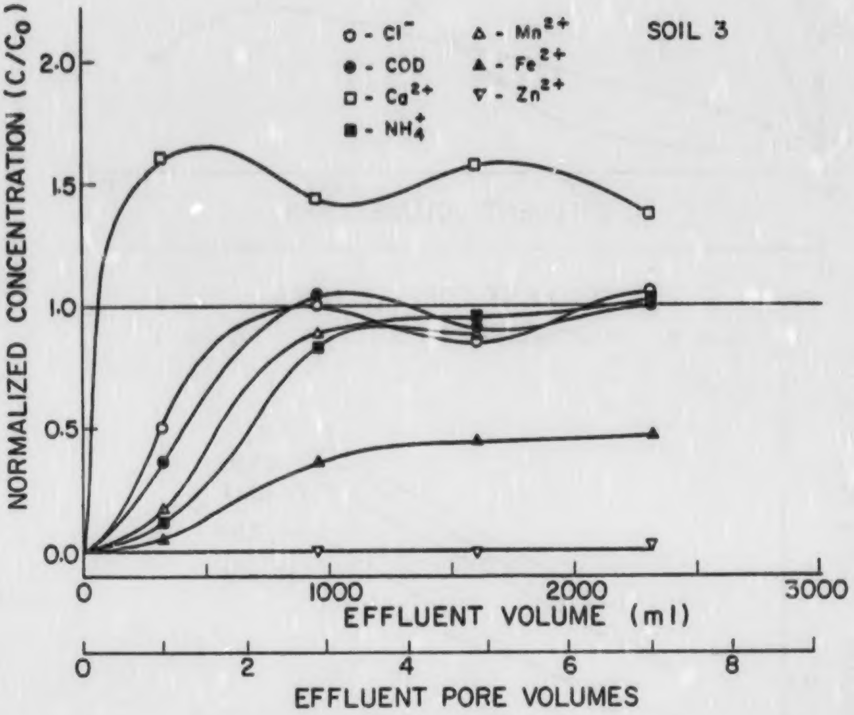

FIGURE 5. BREAKTHROUGH CURVES FOR SOIL 3, COL 1

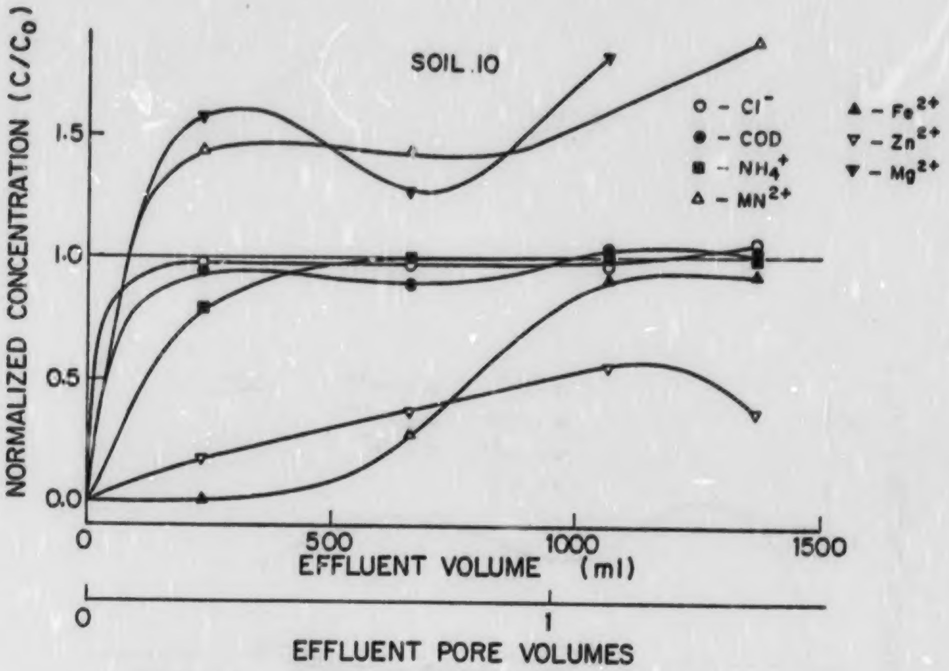

FIGURE 6. BREAKTHROUGH CURVES FOR SOIL 10, COL 1

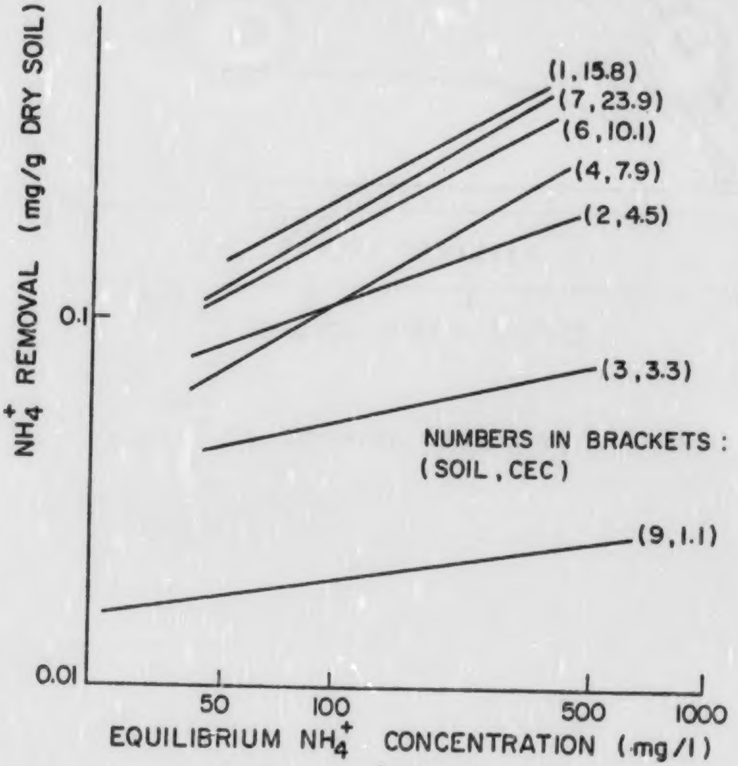

FIGURE 7. NH_4^+ REMOVAL ISOTHERMS

TABLE 3. COMPARISONS OF CONTAMINANT REMOVALS BETWEEN SOILS

Parameter	Contaminant Removed - mg/g For Soil Number 1	 2	 3	 10
COD	.830[1] (28708)	1.01 (28788)	.63 (31490)	.492 (35143)
Fe^{2+}	.052 (94)	.062 (92)	.028 (71)	.018 (55)
K^+	.238 (520)	.132 (532)	-.007[2] (559)	-.117 (503)
Mn^{2+}	.016 (29)	.009 (28)	.002 (27)	-.006 (33)
Ca^{2+}	-.045 (320)	-.085 (300)	-.299 (630)	-.543 (253)
Mg^{2+}	-.141 (195)	-.027 (168)	-.044 (118)	-.03 (215)
NH_4^+	.289 (862)	.224 (846)	.086 (915)	.028 (1030)
CEC meq/100g	15.8	4.5	3.3	1.4
% < 0.002 mm	7.0	8.0	1.7	3.1

[1] number in brackets is leachate concentration

[2] negative sign signifies contaminant release

The dispersed soil reactors (DSR) were used in addition to the soil columns to explore contaminant removal in soil. Isotherms for contaminant removal as a function of concentration were prepared from the DSR data and are presented in Figures 7, 8, 9 and 10 for NH_4^+, Zn^{2+}, COD and K^+, respectively. Data points were omitted for the sake of clarity in the figures. Scatter in the data points was substantial for some soils.

Figure 7 shows NH_4^+ removal isotherms plotted in the Freundlich form. These show a strong trend toward increased removal with increased CEC. The existence of a sorptive process, probably ion exchange, was assumed to exist. A similar trend appears in the data for Zn in Figure 8.

The mechanism of ion exchange would appear to have been operative once again although consideration to precipitation is given subsequently.

In Figure 9, the data for COD are linear on a simple arithmetic plot. Data on COD concentrations less than 12,000 mg/l were not available for Soil 6. Curves for soils 3,9 and 10 with CEC's of 3.3, 1.1 and 1.4 meq/100 gm respectively exhibited limited COD removal in the DSR's. As with NH_4^+ and Zn^{2+} removal, a trend of increased COD removal with increased CEC suggested an adsorptive process as the operative mechanism of removal. The removal of K^+ during leachate

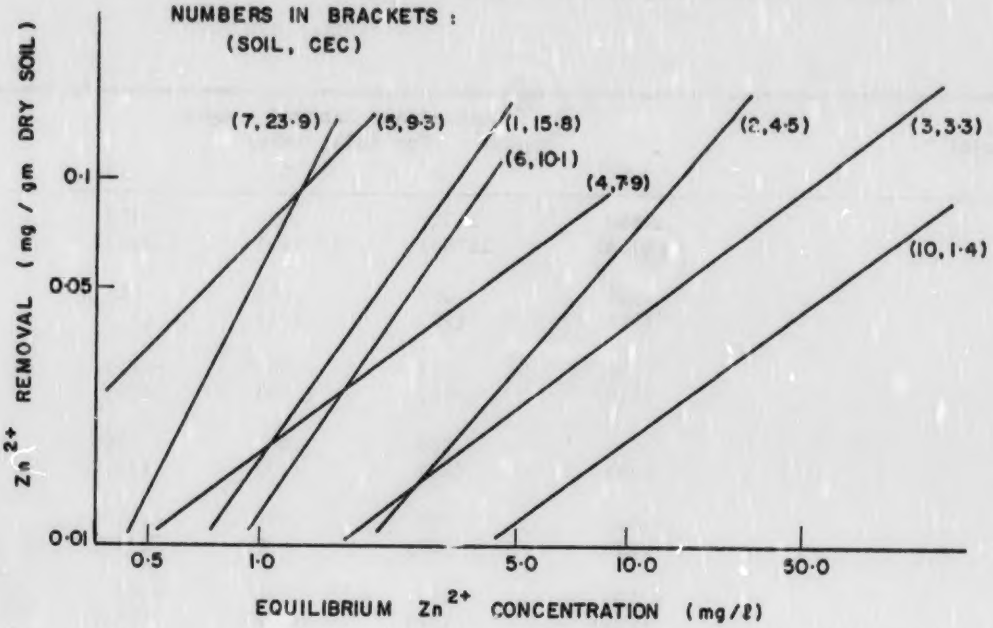

FIGURE 8. Zn^{2+} REMOVAL ISOTHERMS

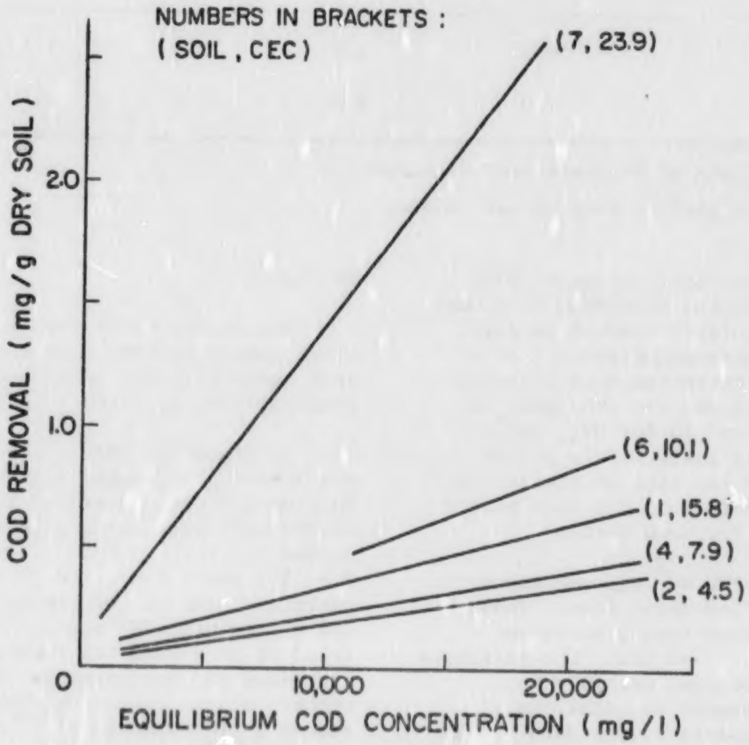

FIGURE 9. COD REMOVAL ISOTHERMS

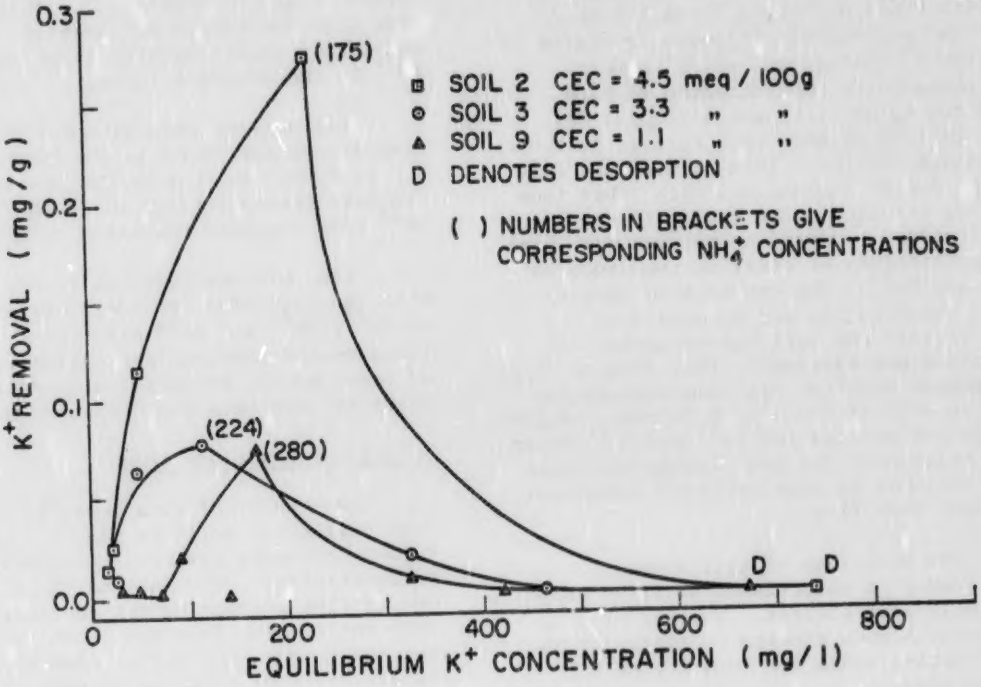

FIGURE 10. SORTION OF K^+ ON SOIL IN OSR

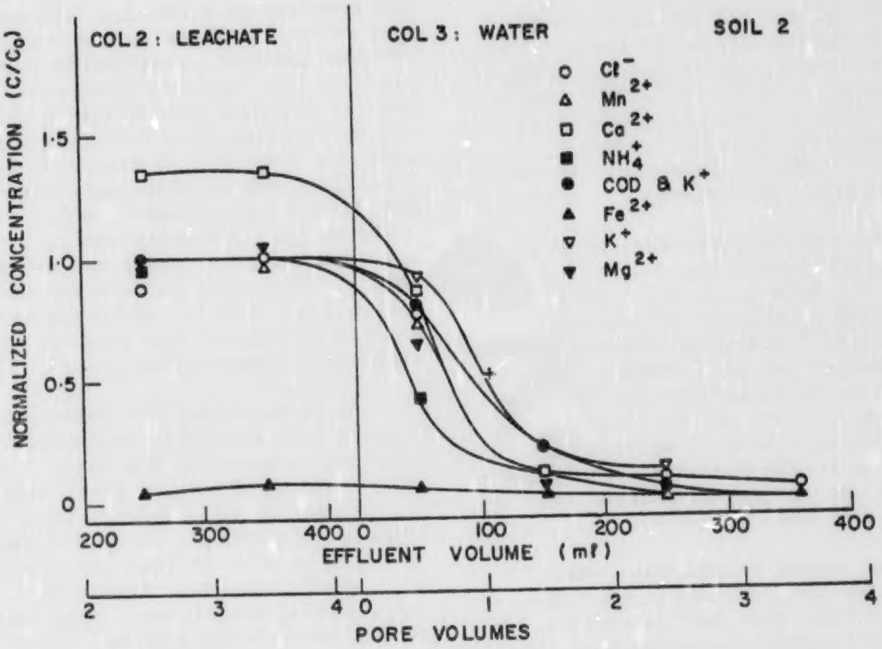

FIGURE 11. CONTAMINANT CONCENTRATIONS DURING DESORPTION

passage through DSR's for soils 2, 3 and 9 with CEC's of 4.5, 3.3 and 1.1 meq/ 100 gm respectively is shown in Figure 10. At low K^+ concentrations, K^+ removal increased with concentration to some maximum value after which removal fell off rapidly to zero with further increases in concentration. This pattern was felt to be due to competition with other ions during sorption. The higher K^+ concentrations occurred coincident with high concentrations of other cations such as NH^+ and Fe^{2+}. The curves show that the NH_4^+ concentration had reduced from 900 to near 200 mg/l before maximum K^+ sorption was observed. This form of interdependent sorption was demonstrated for Na^+ as well as for K^+. A somewhat similar situation existed for Ca^{2+} and Mg^{2+} since the release of the ions during exchanges was followed in some soils by subsequent removal down flow.

The modelling of this interdependency in contaminant sorption is a complicated process. The DSR experimental mode appears to represent an instructive means for quantifying the phenomenon.

Precipitation

Griffin et al. (1, 10) have shown that, under the conditions of their experiments, precipitation was operative in the removal of Pb^{2+}, Zn^{2+}, Hg^{2+}, Cd^{2+}, Cu^{2+} and Cr(IV) at pH's above specified levels. This was supported by the appearance of metallic hydroxides in the surfacial regions of their soil columns.

In this research, evidence of the existance of precipitation was not apparent. The BC's for Soils 1 and 2 (Figures 4 and 2) show no breakthrough for Zn^{2+} and Fe^{2+} and thus, the possibility of precipitation exists. However, breakthrough did occur in Soils 3 and 10. The DSR analysis for both metals strongly suggested the existence of sorptive processes due to the positive correlation between the mass of cation removed and the CEC of the soil (see Figure 8).

Maximum cation solubilities were calculated for the leachate at pH 5.65. It was found that only Zn^{2+} exceeded the calculated solubility (assuming that iron and manganese were present in the divalent form). This was however by a factor of approximately 10^3. This may have been due to the presence of complexing agents helping to maintain Zn^{2+} in the solution phase.

Upon passage through the soil, the leachate pH increased to the range from 7.0 to 7.95. In this range, the measured leachate concentrations of both Fe^{2+} and Mn^{2+} also exceeded predicted solubilities.

From this analysis, it is apparent that the potential for the precipitation of Zn^{2+}, Fe^{2+} and Mn^{2+} did exist. However, based on the previous discussion, removal of these metals by sorptive processes seems to have been the operative mechanism.

Biological Transformations

The results of this research indicated that biological activity played a minor role in reducing leachate contaminant concentrations. The data in Table 3 showed that organic matter (measured as COD) was removed from the leachate during passage through the soil. However, this appears to have been due largely to sorption of the organics.

For Soils 1 and 2, Table 3 showed COD removals of 0.836 and 1.01 mg/gm respectively. These values were calculated for the complete breakthrough curve thus yielding COD concentrations equal to the raw leachate concentration throughout the soil column (see Figures 2 and 4). From the isotherms in Figure 9, the removal of COD would be approximately 0.8 and 0.5 mg/gm respectively. This would suggest only sorption in Soil 1 and sorption plus some other process in Soil 2. The other process may have been biological. Little additional evidence existed to support the presence of biological degradation.

It is reasonable to anticipate the existance of anaerobic decomposition of leachate organics in the soil. Pohland (13) achieved almost complete destruction of organics during the recycle of leachate through a landfill. Boyle and Ham (14) had similar success when treating leachate by means of anaerobic digestion. Thus, the leachate organic matter is amenable to anaerobic decomposition.

It would seem in this research that the conditions for the development of an anaerobic culture suitable for large scale organic matter reduction did not exist. The problem may have been simply one of insufficient time, first for the development of the anaerobic culture and second for it to achieve appreciable reductions in COD.

Support for this exists in the results from the DSR experiments. Two pulses of leachate were passed in succession through the sequence of 10 DSR's for each soil. The interval between slugs was on the order of 1 week during which time residual leachate remained within the soil. The results for most soils showed an appreciable increase in COD removal from the second pulse as compared to the first. It was concluded that the anaerobic culture had an opportunity to develop during the interval and was available to effect increased COD removal during the passage of the second pulse.

Filtration

The leachate used in this research contained suspended solids in very small concentrations in the order 100 mg/1. These were absent in the column effluent and filtration appeared to be the mechanism of removal.

The potential for cation precipitation was discussed previously. It was anticipated that any precipitates formed would be removed by filtration in the soil near the point of formation. This is compatible with the work of Griffin et al. (1).

Gaseous Exchange

The collection and transportation of leachate and the subsequent contact with soil were accomplished under nitrogen gas in order to maintain anoxic conditions. The partial pressure of CO_2 in the landfill cell was in excess of 0.5 atmospheres (4) and thus, CO_2 release from solution was expected during experimentation. Measurements to support this were not made beyond the monitoring of changes in alkalinity and pH.

No other gaseous release was anticipated.

MECHANISM STABILITY

In order to test the stability of leachate contaminant retention in the soil, water was passed through all soil columns and DSR's and contaminant concentrations in the effluents from these experiments were measured. The results from the Soil 2 column desorption experiment are shown in Figure 11.

Once again, Cl^- concentrations were used to indicate ideal tracer behavior. The Cl^- discharge curve reflected the expulsion of leachate remaining in the soil pores from the leachate contact experiments.

The data show that, before the desorption phase, the concentrations of most of the contaminants had reached the raw leachate concentration. Upon desorption, the concentration profiles exhibited the same general shape as the Cl^- curve. The curves for K^+ and COD are of particular interest since they begin at or below $C/Co = 1.0$, but proceed above the Cl^- curve and therefore demonstrate the occurrence of contaminant desorption.

The mass of COD desorbed was calculated to be 0.47 mg/gm. This compares closely with 0.5 mg/gm, the COD sorbed on Soil 2 as estimated from the isotherms in Figure 9. Similar trends were observed in the data from other soils. It would appear therefore that the sorptive processes for COD removal were highly reversible.

Very little desorption of Mn^{2+} or NH_4^+ is evident in Figure 11. The Fe^{2+} curve shows virtually no breakthrough at the end of the leachate contact phase and no apparent release of Fe^{2+} during desorption. The processes responsible for the removal of these ions appear to have been stable during the passage of water through Soil 2.

Desorption was also a part of the DSR experimental programme. A volume of water equivalent to the volume of the leachate pulses was passed through the DSR's in sequence. Effluent contaminant concentrations were measured and the mass of contaminant released calculated. Data for 9 soils are presented in Table 4 for Fe^{2+}, NH_4^+, COD and K^+ for the first DSR only. The influence of dilution in

TABLE 4. COMPARISONS BETWEEN CONTAMINANTS REMOVED BY AND RELEASED FROM SOIL NUMBERS: - (IN mg/gm SOIL)

Contaminant	1[1]	2	3	4	5	6	7	8	9
Fe^{2+}	.132[2]	.235	.022	.204	.146	.246	.161	.185	.021
	-.000[3]	-.004	.004	-.000	-.000	-.000	-.000	-.017	.002
NH_4^+	.607	.220	.117	.646	.054	.449	.490	.150	.012
	-.052	-.030	.018	-.123	-.048	-.203	-.055	.012	-.017
COD	2.022	1.121	1.943	8.696	5.466	-	2.854	-.852	1.243
	-.095	-.838	.630	-1.565	.614	-	-2.157	-.171	-.563
K^+	.356	-.244	-	.447	.253	.471	.295	.037	-.312
	-.053	-.033	-	-.163	-.015	-.073	.038	-.004	.011
(CEC meq/ 100 gm)	15.8	4.5	3.3	7.9	9.3	10.1	23.9	1.1	1.4

[1] data from first DSR in series of 10

[2] leachate contact phase

[3] water contact phase - negative sign indicates desorption

residual soil moisture was subtracted from the raw data.

The upper number is the mass of contaminant released during leachate contact (in mg/gm of soil). The lower number is the mass of contaminant released during desorption. Further contaminant release could be expected with the passage of additional pulses of water and consequently the lower numbers cannot be taken as maxima.

It can be seen that the release of Fe^{2+} during desorption was negligible for all soils. The removal process was therefore highly resistant to the desorption condition.

Such was also the case for NH_4^+ in the majority of the soils. However, in soils 5, 6 and 10, a large percentage of the NH_4^+ removed during leachate contact was released. In Soils 5 and 10, removal during leachate contact had been small.

In most cases COD release during desorption was substantial. This supports statements made earlier in connection with the reversibility of the COD removal process. The behaviour of K^+ during leachate contact varied from 0.471 mg/gm removed in Soil 6 to 0.312 mg/gm released to the liquid from Soil 10. This supports previous conclusions drawn from the K^+ removal isotherms regarding retardation due to other ions in solution.

The trends in contaminant release observed in this research were summarized as:

slight release:	Fe^{2+}, Zn^{2+}, and Mn^{2+}
slight to moderate release:	NH_4^+ and K^+
substantial release:	COD and Na^+

It was noted that Ca^{2+} and Mg^{2+}, while actively released during leachate contact, were in fact removed during desorption experiments (Figure 11).

The results of the desorption experiments show that, for many of the contaminants removed during leachate contact, release during desorption was limited. The resultant concentrations in the desorbing water were substantially less than the initial leachate concentrations. The most active desorption occurred with COD and Na^+.

Such behaviour supports the existence of sorptive removal processes including ion exchange. The very limited release of Fe^{2+}, Mn^{2+} and Zn^{2+} during desorption indicates high selectivity in the exchange sequence.

THE ROLE OF DSR'S

Assessment of soil-contaminant interactions can be accomplished in reduced time using DSR's as opposed to soil column experiments. This is an important factor if such information is required as a prelude to landfill design.

The disadvantages of the DSR's are that they do not simulate the hydraulic conditions of leachate migration in the field in particular dilution and dispersion. Neither do they allow for the development of the microbial population that would occur in situ. Their application appears to be in providing relationships such as sorption isotherms for non-biological contaminant removal mechanisms. The deduction of such relationships from column experiments requires that dilution and dispersion including the chromatographic effect be factored out. At present this is an uncertain activity.

Computer models were prepared for both the column and the DSR experiments to account for the influence of dilution and dispersion on contaminant concentrations. This involved the use of elemental matrix models calibrated by using Cl^- as a tracer. Concentrations during contact were adjusted to reflect reductions due to soil contact only and not dilution and dispersion. The models were run for Soils 1, 2, 3, 9 and 10 and the results presented Figures 12, 13, 14, 15 and 16, respectively. Contaminants chosen for comparison were Cl^-, COD, NH_4^+ and Zn^{2+}. Normalized concentrations were plotted against the soil: leachate contact ratio, gm of soil per ml of leachate. The complete column data are presented as are the data points for the first 4 DSR's.

The data show generally good agreement between the 2 experimental modes. They tend to endorse value of DSR data in providing relationships describing soil-contaminant interactions for use in modelling in situ behaviour.

CONCLUSIONS

The following conclusions were drawn from the results of the column and DSR experiments:

1. Contaminant concentrations at the leachate front were reduced through the mechanisms of dispersion and dilution. These could function effectively with the additions of small pulses of contaminant to the soil.

2. Sorption was the most active contaminant attenuation mechanism observed in this research. Trends of increased contaminant removal in soils of increased CEC were demonstrated for NH_4^+, K^+, Fe^{2+}, Zn^{2+}, Mn^{2+} and COD. Ion balances showed that sorption processes in addition to cation exchange were operative.

3. Increased pH during soil contact created the potential for Fe^{2+}, Mn^{2+} and Zn^{2+} precipitation.

4. Limited removal of organic matter by microbial activity was demonstrated.

5. Desorption studies demonstrated only slight release of Fe^{2+}, Zn^{2+} and Mn^{2+} to water, slight to moderate release of NH_4^+ and K^+ and substantial release of COD and Na^+.

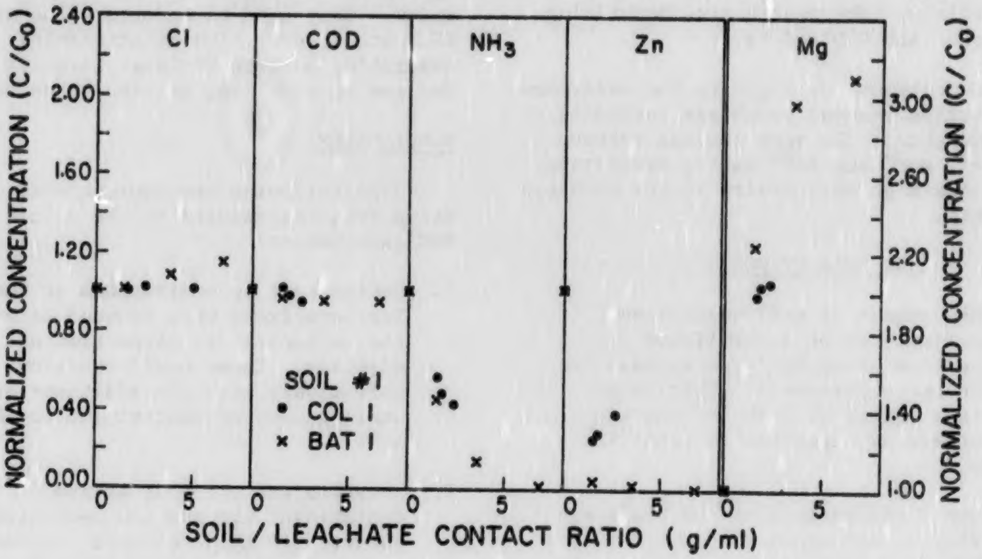

FIGURE 12. COMPARISONS BETWEEN COLUMNS AND DSRS: SOIL 1

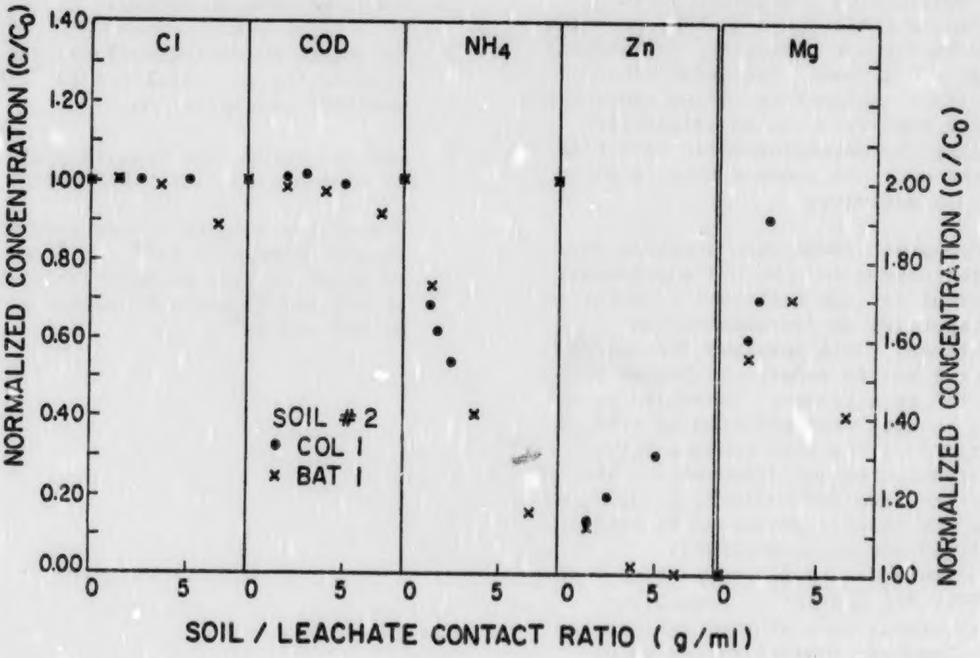

FIGURE 13. COMPARISONS BETWEEN COLUMNS AND DSRS: SOIL 2

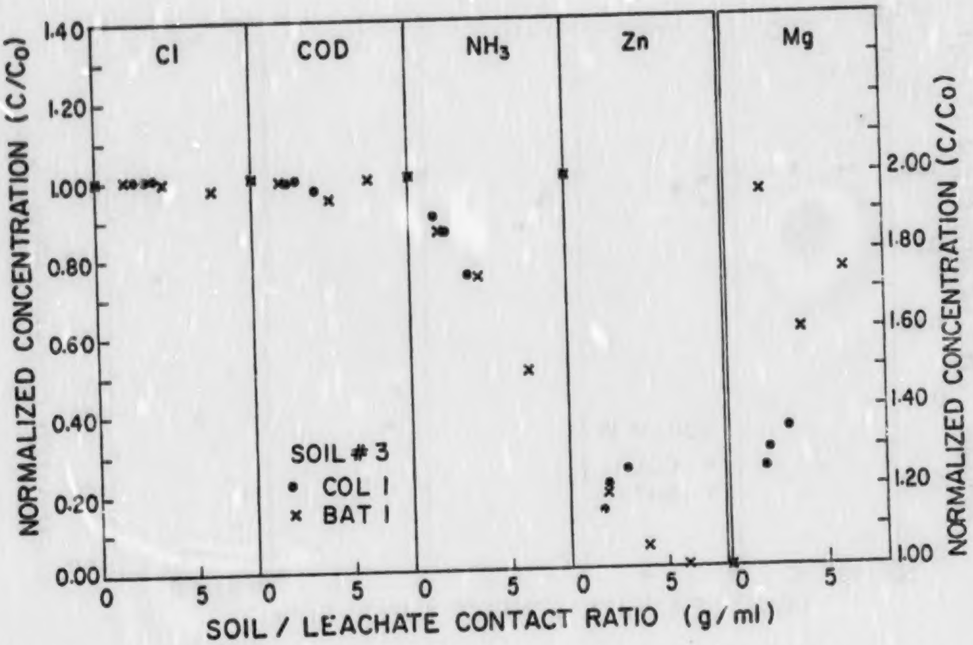

FIGURE 14. COMPARISONS BETWEEN COLUMNS AND DSRS: SOIL 3

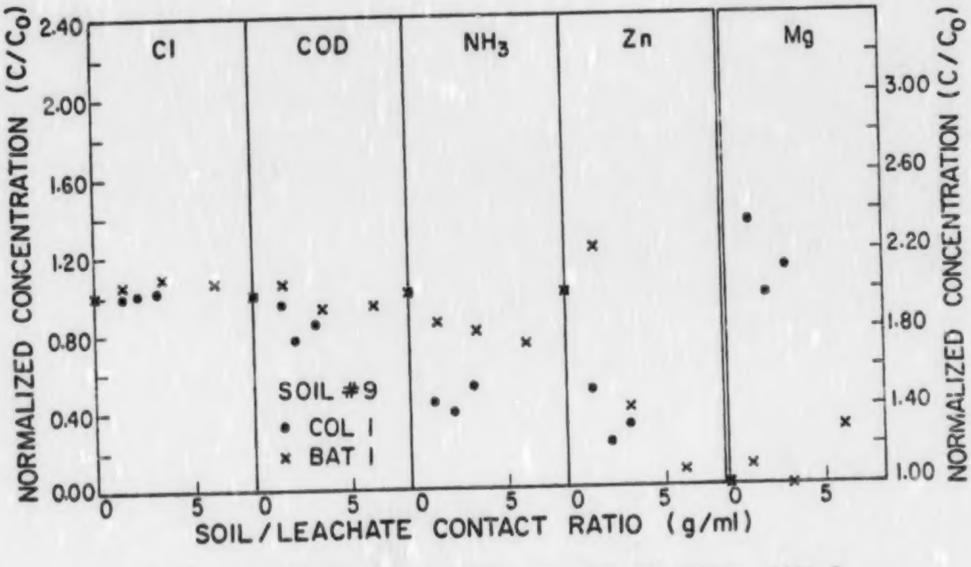

FIGURE 15. COMPARISONS BETWEEN COLUMNS AND DSRS: SOIL 9

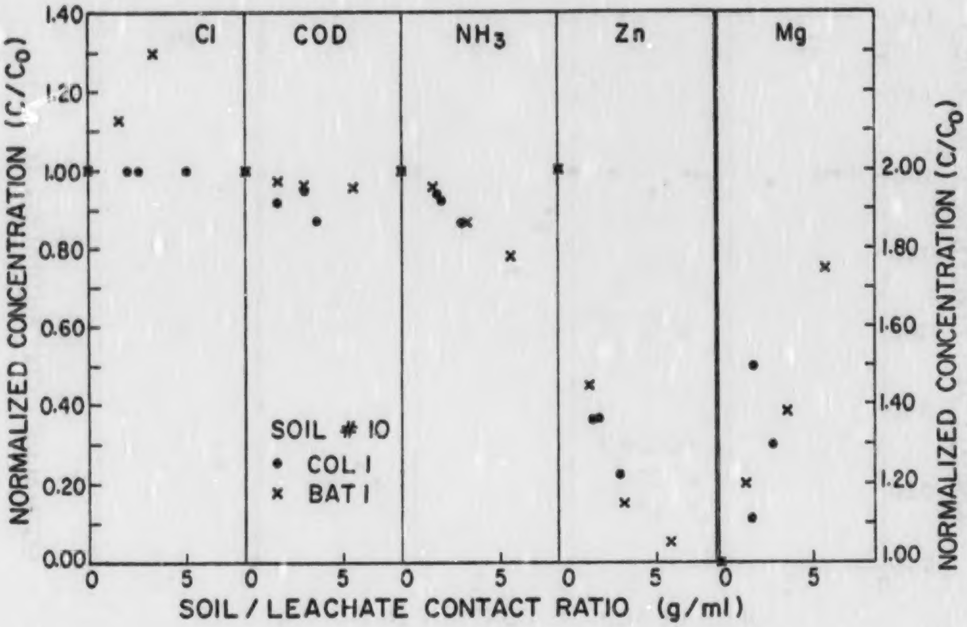

FIGURE 16. COMPARISONS BETWEEN COLUMNS AND DSRS: SOIL 10

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ATTENUATION OF PCB'S BY SOIL MATERIALS AND CHAR WASTES

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ABSTRACT

Adsorption of polychlorinated biphenyl (PCB) isomeric mixtures containing 42 and 54 percent chlorine by montmorillonite clay and soil and the relative mobility of these compounds through soil media were determined by both gas chromatography and ^{14}C labeling techniques. Adsorption by these earth materials was found to be strong with more than 90 percent removal from solution at concentrations approaching the water solubility of the compounds tested.

PCB's were found to be immobile in earth materials when measured by the soil thin-layer chromatography technique. R_f values for PCB's were found to be zero to 0.02 for all amounts of PCB's tested (42-206 ng). Dicamba, a pesticide with high mobility, was used as an internal standard and yielded R_f values of 0.80 to 1.00.

Gas chromatographic analytical procedures that allowed improved quantitative measurement of PCB's in aqueous solutions were developed. The overall perchlorination procedure for conversion of isomeric mixtures of PCB's to the fully chlorinated biphenyl by digestion with $SbCl_5$ was successfully reduced from approximately 20 steps to about 10 steps. The speed of the analyses was improved and interference from bromine was removed. Reproducibility of the overall perchlorination with 80 ng biphenyl in sealed glass tubes was determined to be 0.52 percent relative standard deviation.

INTRODUCTION

Polychlorinated biphenyls (PCB's) are used in a wide range of industrial applications such as electrical insulation, fire-resistant and heat transfer fluids, hydraulic fluids, high temperature and pressure lubricants, sealants, expansion media, adhesives, plasticized paints, lacquers, varnishes, pigments, paper coatings, waxes, and as constituents in elastomers. They were largely ignored as environmental contaminants until Jensen (1) and Widmark (2) identified them in 1966. PCB's did not attract much concern as hazardous chemicals until the incidents of contaminated cooking oil in Japan in 1968 and of contaminated chicken feed in the United States in 1971 (3). Laboratory studies with animals have shown that PCB's can cause enlargement of the liver, induction of hepatic microsomal enzymes, reproductive failures, gastric disorders, skin lesions, and tumors in birds and mammals (3). The 2000 afflicted Japanese people in the "Yusho" incident of 1968 experienced lesions of the skin, facial swelling, and neurological disorders that were similar to the results reported in the animal studies (4).

Fish and other aquatic organisms tend to accumulate PCB's in lipid-rich tissues and organs. Predators at the top of the food chain may accumulate PCB's to levels of more than 10^7 times that of the ambient water (4). Man usually resides at the top of the various food chains and, due to the biological magnification, may ingest large amounts of PCB's even though only trace amounts are present in the ambient waters.

PCB's have, therefore, been considered as a significant hazard to human health as well as the environment.

PCB's have been manufactured in the United States since 1929; it has been estimated that more than 400,000 tons have been produced since that time. The sole U.S. manufacturer of PCB's is the Monsanto Company located near East St. Louis, Illinois. Since 1971, Monsanto voluntarily has restricted its sales of PCB's to only "closed" systems, such as PCB-containing insulating fluids used in electrical transformers and capacitors. These two applications account for essentially all the current use of PCB's in the United States (5). On October 5, 1976, Monsanto announced that it would cease to manufacture and distribute PCB's by October 31, 1977. A timetable set by the U.S.-EPA has called for a gradual phasing out of PCB manufacturing by January 1, 1979, and a ban on all PCB processing or distribution in commerce by July 1, 1979 (6). These steps have significantly reduced the introduction of PCB's into the environment.

Unfortunately, approximately one-half million pounds of PCB's are still imported into the U.S. each year from foreign manufacturers and millions of pounds of PCB's still exist causing the environmental levels to remain quite high. For example, two tributaries of Lake Michigan have PCB levels that consistently exceed 100 ppt. This has contributed to PCB levels between 4 and 10 ppt in certain parts of the lake. The present U.S.-EPA recommended water quality criteria is less than 1 ppt and the high PCB levels have caused great concern to the residents of Chicago, who draw their drinking water from the lake (4).

Many companies discard their old electrical equipment in unapproved places and thus discharge the PCB's into the atmosphere and waterways. One problem of disposal involves the high costs and fees for transporting PCB wastes to regional incinerators or approved landfills vs. simply discarding the wastes. Incineration is considered the safest method for disposal of PCB wastes. However, this method is extremely costly and has some operating difficulties. PCB's do not burn readily and, under improper operating conditions, can be vaporized during incineration. Thus, incineration may turn out to be the major source of PCB's reentering the environment. In addition, large electrical transformers and capacitors, the major source of waste PCB's, cannot be satisfactorily incinerated.

Thus, land disposal is the only reasonable alternative for waste PCB's. Although landfill disposal appears to be the most acceptable alternative, little information is presently available concerning the possibility of ground-water contamination by leaching PCB's from landfills. Lidgett and Vodden (7) analyzed waters around a sanitary landfill for PCB's and found the contamination levels to be below their detection limit of 4 ppb. Similarly, Robertson and Li (8) failed to detect PCB's in ground water using GC/Mass Spectrometry techniques. Tucker, Litschgi, and Mess (9) studied the leaching of Aroclor 1016 from various types of soils and concluded that PCB's are not readily leached from soil by percolating water.

The paucity of information available shows no evidence that ground waters have become contaminated by PCB's. However, many surface waters do contain PCB's and the mechanism of transport in the biosphere and the mechanism of attenuation in soil are still unknown. Data on the factors affecting PCB attenuation by earth materials would provide a rational basis for future disposal site selection and design.

BACKGROUND

The research reported here is supported in part by Grant R-804684-01, from the U.S. Environmental Protection Agency, Municipal Environmental Research Laboratory, Solid and Hazardous Waste Research Division, Cincinnati, OH 45268.

The purposes of the present project are:
a) To conduct an extensive literature review of pertinent information on the adsorption of hazardous organic compounds;
b) To measure the adsorption capacity of selected earth materials for pure PCB's and PCB wastes;
c) To quantitatively evaluate the effects of pH, biological degradation, photodecomposition, volatilization, time, and adsorbent structure on adsorption of PCB's;
d) To use this data to develop a mathematical model that will allow prediction of PCB adsorption and mobility; and
e) To further develop analytical procedures that will allow improved quantitative measurement of PCB's contained in aqueous solutions.

PCB Materials

Polychlorinated biphenyls (PCB's) is a generic term applied to certain mixtures of synthetic organic compounds. These compounds are mixtures of very closely related isomers and homologs that contain two phenyl rings with 10 possible chlorine attachments. The biphenyl structure is shown in Figure 1. PCB's are made by substituting chlorine atoms for one or more of the hydrogen atoms at the numbered positions of the biphenyl structure. These compounds are chemically and thermally stable, very resistant to microbial degradation, and are highly persistent in the environment.

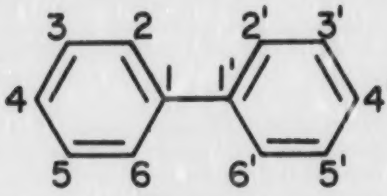

Fig. 1. BIPHENYL STRUCTURE: Positions 2 to 6 and 2' to 6' indicate ten possible positions for chlorine substitution. Different amounts of chlorine substitution form the various PCB's.

The PCB materials chosen for study were the pure Aroclors 1242 and 1254 (42 and 54% substituted chlorine, respectively) supplied by the Monsanto Company, and the ^{14}C labeled compounds were prepared by New England Nuclear Corporation. Gas chromatographic traces of the ^{14}C labeled compounds were identical to those of the pure Aroclors 1242 and 1254, respectively. Therefore, it was assumed that there were no significant differences in the respective compounds and that the ^{14}C labeled and pure Aroclors would behave similarly in studies of adsorption, mobility, and microbial degradation.

A used capacitor fluid was also obtained for study. The fluid was drained from a burned out 50 KVA capacitor manufactured by Westinghouse in 1966 and originally contained Aroclor 1242. This capacitor was supplied by Illinois Power Company and was scheduled to be landfilled. We believe this fluid is representative of the type of PCB wastes that are normally disposed of in landfills.

Adsorbents

Earth materials, representing a wide range in characteristics, have been selected as adsorbents. The materials being studied are: Ottawa silica sand; Panther Creek southern bentonite clay; the soils, namely Bloomfield ls, Ava sic, Cisne sil, Flanagan sil, Catlin sil, Drummer sicl, Weir sic, a calcareous loam till; and two coal chars. The chars were selected because of their high adsorption capacity for organic compounds. They are a waste product of many coal conversion processes and thus have potential use as a liner material for disposal sites accepting organic wastes.

Analytical Development

In general, PCB's are determined quantitatively by comparing gas chromatographic (GC) response patterns of a multicomponent environmental sample with commercial PCB's (Aroclors) or a mixture of Aroclors. This technique is limited by the sensitivity and reproducibility of comparisons of the large number of peaks produced by the various PCB isomers. The procedure is further complicated because the various components of water soluble PCB's contained in environmental samples are not likely to have the same composition as those in the original Aroclor used as a reference compound. For practical reasons, the quantitation is usually done by integration of the major peaks while ignoring the minor peaks. This can cause some error, depending on how well the mixture of isomers in an unknown sample compares to a standard.

Because of these problems, we have developed procedures that allow improved quantitative measurement of PCB's in aqueous samples. The main thrust of our studies has been to improve previous procedures whereby isomeric mixtures of PCB's were converted to the fully chlorinated biphenyl, decachlorobiphenyl (DCB), by digestion with $SbCl_5$. This procedure has the advantage of converting all the PCB's to a single peak for improved quantitation. The electron capture GC detector is many times more sensitive to DCB than it is to PCB's; thus, the conversion to DCB improves the sensitivity and lowers the detection limit for PCB's.

CURRENT STUDIES OF PCB ATTENUATION AND ANALYTICAL DEVELOPMENT

PCB Mobility

The technique of determining pesticide mobility in soils by soil thin-layer chromatography was introduced in 1968 by Helling and Turner (10). Since the introduction of the technique, the mobility of a large number of pesticides in a variety of soils has been tested (11, 12, 13). Soil thin-layer chromatography, or soil TLC, is a laboratory method that uses soil as the adsorbent phase and water as the developing solvent in a TLC system. The system is relatively simple and yields quantitative data on the mobility of organic compounds in soils that appear to correlate well with trends noted in the literature (10). The results reported here are mobility data for Aroclor 1242 and 1254 on TLC plates made from sand, clay, three soils, and a coal char. Dicamba, a pesticide of known high mobility, was used as an internal standard.

The soil sample was slurried with water until moderately fluid, and then was applied with a spreader to clean glass plates 20 cm by 20 cm square. The soil was spread to a thickness of 0.5 mm and then air dried. A horizontal line was scribed 12 cm above the base to stop water movement; vertical lines were scribed 2 cm apart to separate the various treatments. The compounds were spotted 2 cm from the base and leached 10 cm with water. The activity of the ^{14}C labeled compound varied between 11,000 and 44,000 dpm. The plates were immersed in 0.5 cm of water in a closed glass chamber and were removed when the wetting front reached the horizontal line. Leaching was thus ascending chromatography. The soil plate was then removed and air dried. A piece of 8 × 10 inch medical X-ray film was placed in direct contact with the soil plate for approximately one week. The resultant autoradiograph indicated the relative movement of the compound, which was measured as the frontal R_f of the spot or streak.

Figure 2 shows the results of PCB and Dicamba mobility on Catlin soil plates. The figure is a composite of data from two plates illustrating the low mobility of the two PCB's at four concentrations and the excellent replication of the Dicamba mobility. The amounts of PCB spotted in each lane is labeled on the figure and ranged from 42 to 206 ng. It is clear that at all four amounts PCB's remained at the origin, were immobile in Catlin soil, and the Dicamba had an R_f of between 0.85 and 0.90. The R_f is defined as the distance the compound moved relative to the distance the water front moved; that is, the Dicamba moved 85 to 90 percent of the distance the water front moved on the plate. The two PCB's had R_f values of zero.

The R_f values obtained for Aroclor 1242 and 1254, and for Dicamba on TLC plates made with several earth materials are presented in Table 1. The results clearly indicate that the two PCB's tested are highly immobile in these test systems. R_f values of zero to 0.02 were obtained for all the materials tested, even the pure silica sand. Dicamba was shown to be highly mobile in these tests with R_f values ranging from 0.80 in the char to 1.00 in the sandy materials.

Adsorption Studies

Equilibrium adsorption studies were carried out by shaking known volumes of PCB solutions with varying weights of earth materials at a constant temperature of 25°C. Figure 3 shows representative results for adsorption of Aroclor 1242 and 1254 by montmorillonite clay. Weights of clay varied from 0.01 to 0.5 g per 10 ml of solution. Blanks containing no clay were carried through the experiment. The data in Figure 3 indicates that more than 50 percent of the PCB's were removed in the blanks (no clay). The reaction was carried out in sealed centrifuge bottles so that volatilization and losses during separation of the solid from the liquid phase were minimized. Since PCB's are highly resistant to microbial degradation, the results are interpreted as adsorption of the PCB's onto the glass walls of the centrifuge bottle. This strong adsorption by the glass container is consistent with the observation that PCB's were immobile on the silica sand TLC plates described above. Adsorption by 0.5 g of clay is nearly complete with less than 1 ppb remaining in solution. It was concluded that PCB's are strongly adsorbed by earth materials. This conclusion is consistent with the high degree of immobility observed in the soil TLC study.

Analytical Procedure Development

Little effort has been made to derive

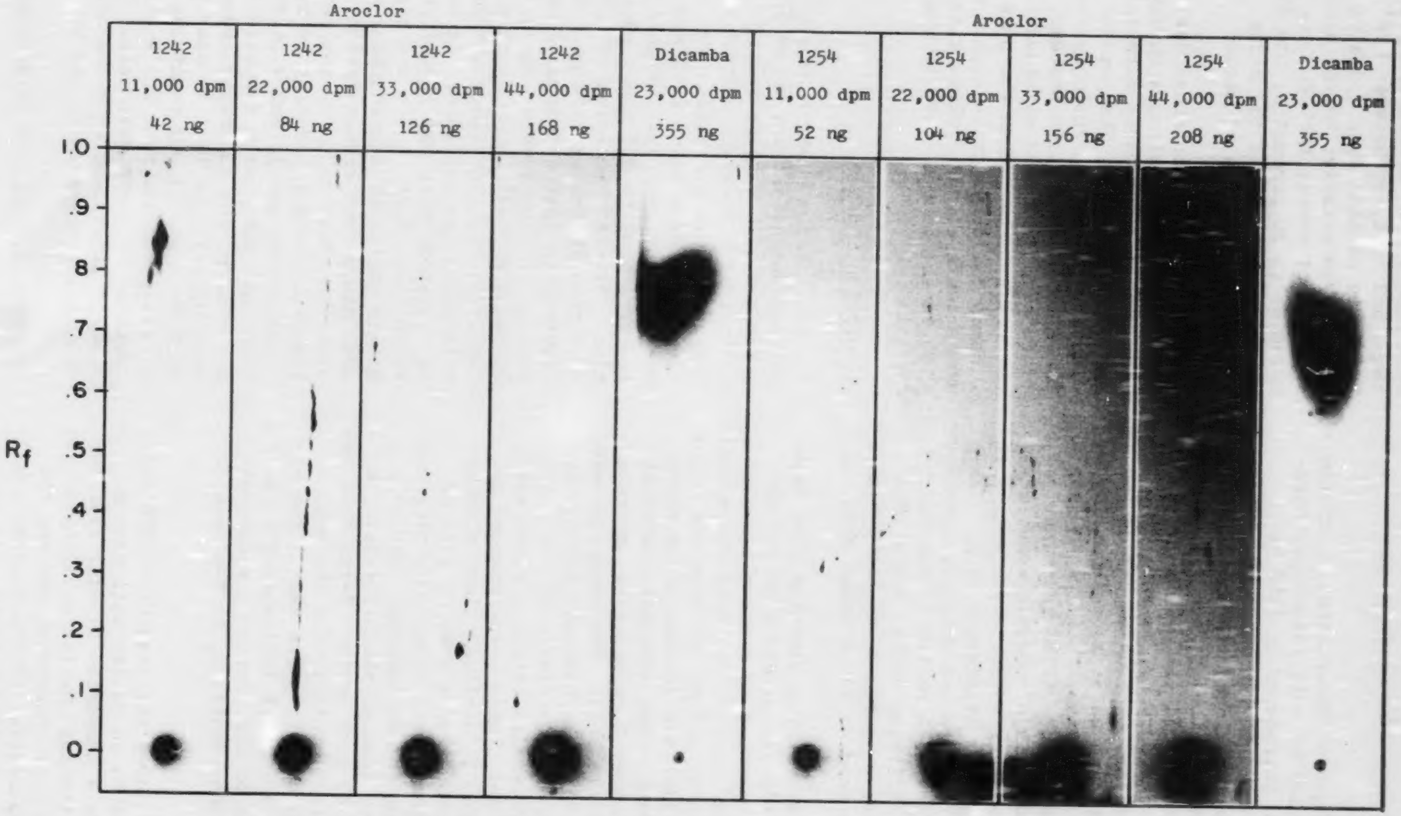
Aroclor
Aroclor
1242
11,000 dpm
42 ng
1242
22,000 dpm
84 ng
1242
33,000 dpm
126 ng
1242
44,000 dpm
168 ng
Dicamba
23,000 dpm
355 ng
1254
11,000 dpm
52 ng
1254
22,000 dpm
104 ng
1254
33,000 dpm
156 ng
1254
44,000 dpm
208 ng
Dicamba
23,000 dpm
355 ng
R$_f$
1.0
.9
.8
.7
.6
.5
.4
.3
.2
.1
0

Fig. 2. PCB and Dicamba mobility in water on Catlin sil soil thin-layer chromatography plates.

Table 1: Mobility of Aroclors 1242 and 1254 and Dicamba in Earth Materials as Measured by Soil Thin-Layer Chromatography.

Earth Material	Compound		
	Aroclor 1242	Aroclor 1254	Dicamba
	R_f		
Silica sand	.02	.02	1.00
Bloomfield ls	.01	.01	1.00
Ava sic	.00	.00	1.00
Catlin sil	.00	.00	.88
Montmorillonite	.00	.00	1.00
Coal Char (1200°F)	.00	.00	.80

the PCB residue to a single compound for quantification. The attempted derivatives were biphenyl and decachlorobiphenyl (DCB). The former derivative could be obtained by catalytic hydrogenation of PCB's (14, 15). The main disadvantage of this approach is that the derivative, biphenyl, is determined with flame ionization detector (FID) on GC, which decreases the sensitivity of the detection system.

Procedures have been established to convert PCB's to perchlorinated PCB, DCB, by Armour (16). In brief, an extracted PCB residue with 0.2 to 0.5 ml antimony pentachloride in ~0.1 ml chloroform was subjected to heating at 170° for 4 to 15 hours. The reaction vessel used was 10 OD mm × 150 mm (internal volume ~7.5 ml) resealable glass tubes; heat was applied to one third of the tube length during the reaction. At the end of the heating the excess $SbCl_5$ was decomposed with 6N HCl, followed by hexane extraction of DCB for GC analysis. Overall perchlorination reaction requires approximately 20 steps.

The procedure has been subjected to further study by Trotter (17) and the limitations on the use of $SbCl_5$ for perchlorination of PCB's were discussed. Commercially available $SbCl_5$ is contaminated with traces of Br, likely as antimonybromotetrachloride. The presence of the Br contaminants is believed to be the source of bromonanochlorobiphenyl (BNCB), which was a likely competing product with DCB during perchlorination.

Based on the above information, our efforts have been directed toward modification of the perchlorination procedure by:
1) Increasing the amount of solvent used during perchlorination,
2) Removing the Br interference,
3) Reducing the overall number of reaction steps.

The gas chromatographic column used in this study consisted of a 2 mm ID × 1.83 m glass column packed with 4% SE 30/6% OV-210 on 80/100 mesh chromosorb WHP. The column temperature was held isothermally at 260°C. The detector used was a Hewlett Packard linear ^{63}Ni electron capture (ECD) with nitrogen as a carrier gas. The attenuation of the gas chromatograph was set at 1 × 4 with which full-scale recorder deflection was observed with 100 ng DCB.

For the gas chromatographic quantification, Mirex was used as an internal standard. Reproducibility of gas chromatographic quantitation of DCB with Mirex as an internal standard will be discussed in the later part of this section.

By increasing the amount of perchlorination solvent, loss of extracted PCB's can be avoided during concentration of the extracts (18). Also, a larger volume of solvent will maintain reflexing inside the reaction vessel resulting in complete mixing of PCB's and the perchlorinating reagent, $SbCl_5$.

When 2 ml of a mixture containing equal amounts of $CHCl_3$ and $SbCl_5$ was heated overnight at a heating block temperature of 220°C, interfering gas chromatographic peaks were observed. The interfering GC peaks may result from a reaction between the $SbCl_5$ and the $CHCl_3$, or between $SbCl_5$

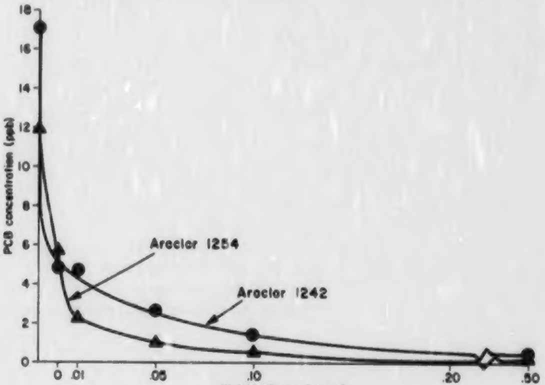

Fig. 3. Adsorption of Aroclor 1242 and 1254 by montmorillonite at 25°C.

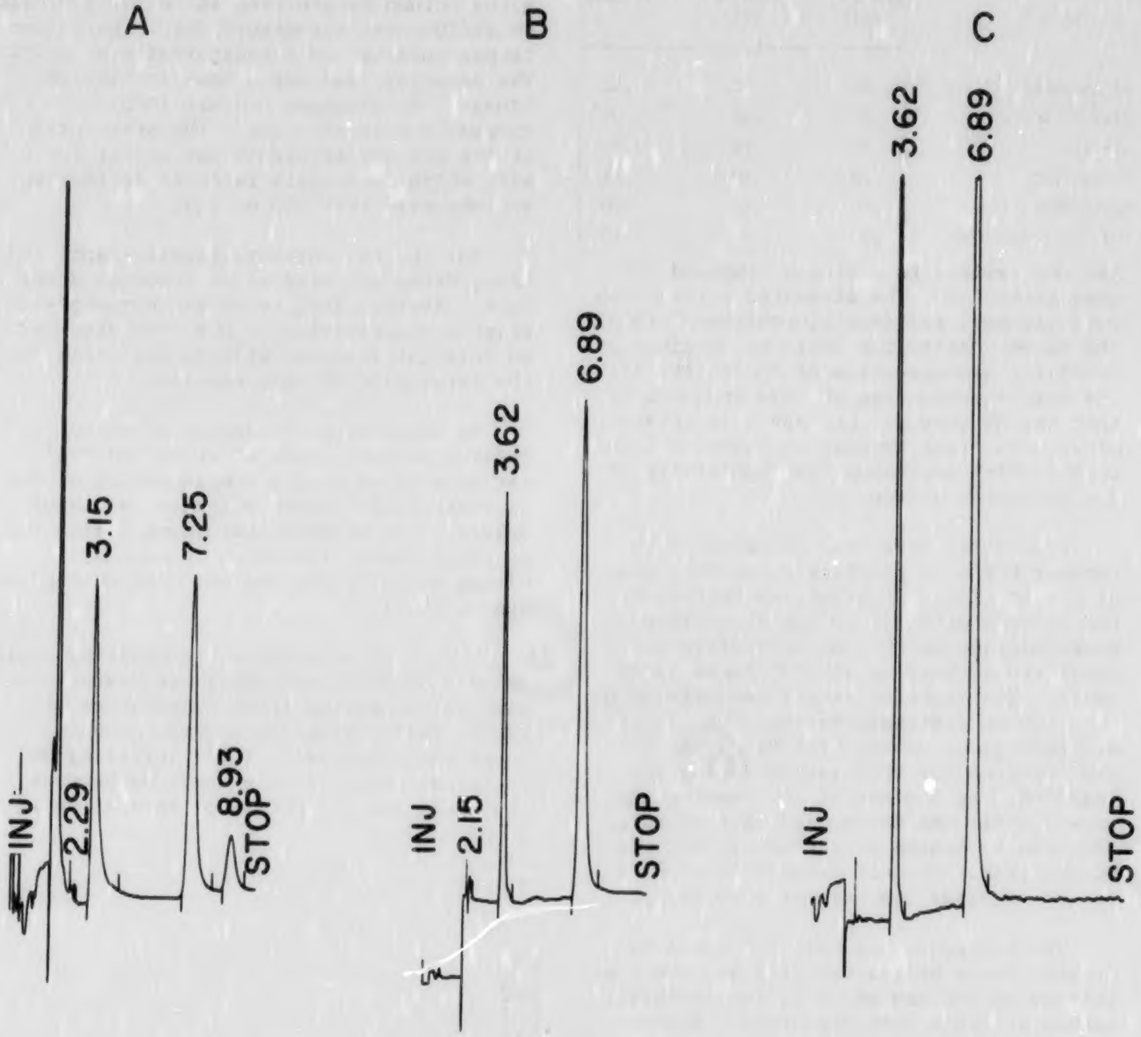

Fig. 4. Gas chromatography traces for PCB analysis. A) Interference caused by Br and incomplete perchlorination. B) Complete perchlorination showing disappearance of Br interference. C) Analysis of actual water sample using perchlorination procedure.

and the C_2H_5OH that is present in $CHCl_3$ as a preservative. Further, the use of CH_2Cl_2 gave a similar result. However, CCl_4 did not react with $SbCl_5$ under the given reaction conditions.

A close examination of the distributions of the reaction products DCB and BNCB reveals that the halogenation steps are reversible. When a mixture of 0.5 ml of $SbCl_5$ and 80 ng of biphenyl in 1 ml of CCl_4 was reacted at 220°C for different periods of time, the distribution of these products also changes. BNCB is believed to be formed from the reaction between biphenyl and $SbBrCl_4$, which is present as an impurity in $SbCl_5$. The ratios of DCB and BNCB vs. reaction times are given in Table 2.

Table 2: Effect of Reaction Time on Formation of DCB and BNCB.

Reaction time	% of DCB Formed	% of BNCB Formed
4 hours	83	17
8 hours	93	7
25 hours	96	4

It is clear that a longer reaction time favors the formation of DCB over BNCB and indicates that the initial formation of BNCB is a kinetically controlled reaction. Further reaction time shifts the unstable BNCB to the more stable compound DCB—that is, it is a thermodynamically controlled reaction. Figure 4A shows that when the reaction is incomplete the BNCB peak appears at 8.6 on the GC trace. When the reaction is complete, BNCB does not appear on the GC trace (Fig. 4B). Thus interference from Br can be eliminated by complete reaction in the perchlorination step. Furthermore, the appearance of the BNCB peak indicates that complete perchlorination is not achieved in the particular sample.

The overall perchlorination procedure of PCB's was successfully reduced to approximately 10 steps by modifying the usual methods of liquid-liquid extraction and evaporation of the solvent. This new technique now only requires 15 to 20 minutes to prepare a finished sample after perchlorination of PCB's. The detailed procedure is shown in Figure 5. It was found that the sample reaction temperature must be at least 180°C for more than 16 hours to achieve complete reaction and destruction of BNCB. In our studies, this procedure required a heating block temperature of 240°C. Figure 4C shows a typical GC trace for a water sample processed by the procedure shown in Figure 5.

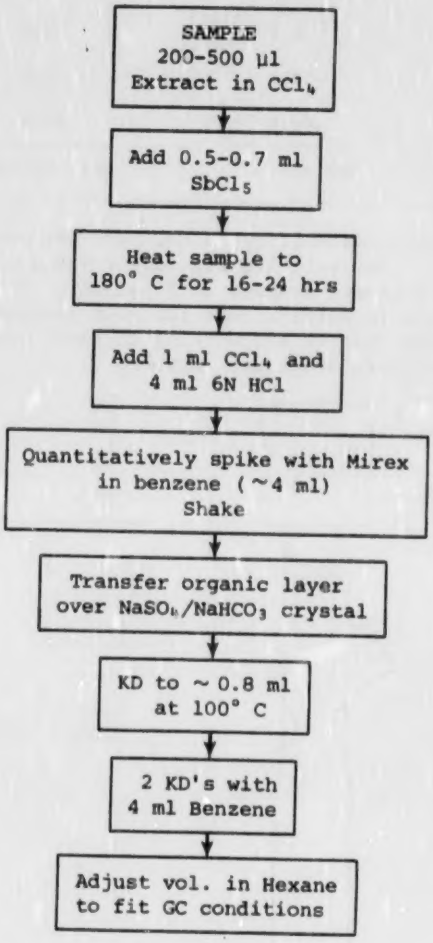
PCB ANALYTICAL PROCEDURE
SAMPLE
200-500 µl
Extract in CCl_4
Add 0.5-0.7 ml $SbCl_5$
Heat sample to 180° C for 16-24 hrs
Add 1 ml CCl_4 and 4 ml 6N HCl
Quantitatively spike with Mirex in benzene (~4 ml) Shake
Transfer organic layer over $NaSO_4/NaHCO_3$ crystal
KD to ~ 0.8 ml at 100° C
2 KD's with 4 ml Benzene
Adjust vol. in Hexane to fit GC conditions

Fig. 5. Schematic block diagram of procedures used in PCB analysis by perchlorination with $SbCl_5$.

The reaction step was initially carried out in glass sealed tubes and excellent reproducibility was obtained. Table 3 shows representative results from four runs where a relative standard deviation of 0.52 percent was obtained. However, the glass sealed tubes were subject to explosion and created a safety hazard to workers in the laboratory. Therefore, the glass tubes were replaced with the teflon plugged reaction tubes shown in Figure 6; the teflon plugged tubes were more convenient and safer to use. However, Table 3 shows that

Table 3: Perchlorination of Biphnyl to DCB.

Sample	Glass Sealed	Teflon Sealed
1	58.7	69.1
2	58.8	72.8
3	59.0	74.8
4	59.4	66.8
	RSD = 0.52%	RSD = 5.0%

much more variability in the data was obtained. The relative standard deviation from replicate samples is 5 percent. This accuracy is satisfactory for most routine analysis, but we are working to find ways to improve this procedure further.

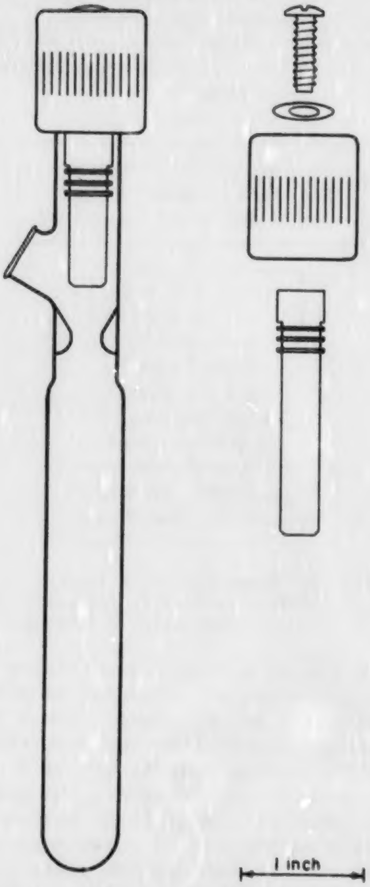

Fig. 6. Teflon plugged reaction vessel used in perchlorination reaction.

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VEGETATION KILLS IN LANDFILL ENVIRONS

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ABSTRACT

Much of the organic matter in refuse landfills biodgrades anaerobically producing primarily methane (CH_4) and carbon dioxide (CO_2). The death of vegetation above and adjacent to buried refuse has been associated with the presence of these gases in the soil atmosphere.

Data obtained from surveying a number of landfill vegetation sites throughout the United States are reported. Preliminary indications are that the death of the vegetation is associated with anaerobic soil conditions caused by the landfill gases. Methods for evaluating the potential for landfill vegetation growth problems are presented along with suggested control measures to reduce these vegetation losses.

Landfill Gas Generation and Movement

Prior to the 1960's many landfills were frequently operated as open-burning dumps. While this caused air pollution and vector control problems at the time of their operation, the end result was that most of the material left in the landfill was non-biodegradable. This meant that there was much less ultimate settlement of the landfill and much less or practically no development of anaerobic gases after the landfill was closed. The modern landfill does not permit open burning; therefore, it provides much more food for microorganisms. It is these microorganisms which generate the gases, mainly carbon dioxide and methane, which present problems for growing vegetation.

When the refuse is first placed in the landfill it is intermixed with oxygen. Therefore, the initial decomposition takes place aerobically, resulting mostly in the generation of water vapor and carbon dioxide. However, this oxygen is soon consumed, and since the refuse is deposited with compaction and covered with soil, there is minimal opportunity for new air to move into the refuse. This results in anaerobic degradation of the refuse, whereby methane and carbon dioxide are produced.

High concentrations of carbon dioxide in the root zone of plants has been reported to be directly toxic to the vegetation (1,2). Although methane has not been reported to be toxic to the vegetation per se; it is possible that, combined with the carbon dioxide it can remove oxygen from the root zone of the vegetation by direct displacement and/or by utilization of the oxygen by methane-consuming bacteria. The end result is a very low or zero oxygen tension in the root zone of the vegetation, which could be toxic to the vegetation (3). In addition, there are frequently minor fractions of a number of other gases present in the landfill gases (4) including ammonia, hydrogen, hydrogen sulfide, mercaptans, ethylene, etc. One or more of these gases have been known to exhibit direct toxicity to vegetation. In any case, we associate the presence of persistent high concentrations of combustible gases in the soil atmosphere with poor health and eventual demise of vegetation.

In addition to causing vegetation growth problems, methane is combustible. This can result in fires and/or explosions where the gas reaches its flammable limits (5 to 15% methane in air) in confined spaces, such as a sump, a room, or a building.

Many landfills are being built into the ground to depths greater than 50 ft. This extensive depth of refuse coupled with the high compaction and daily cover can result in high gas pressures building up within the landfills, at times greater than 5 pounds per square inch (5). These gases travel out of the landfill via the easiest route. This can result in the landfill gases traveling beneath the ground laterally from the landfill. Land adjacent to former sand and gravel pits is particularly receptive to this migration. The landfill gases may migrate to areas where they can cause poor health and/or death of vegetation. They can also migrate into buildings where they may cause fires and/or explosions. Landfills in former sand and gravel pits are particularly noted for having problems with the external migration of landfill generated gases into the surrounding soils. This is in most cases due to the high porosity of adjacent soils and their lack of resistance to the movement of these gases. We have also observed that when the mixtures of adjacent soils are horizontal layers of sand and gravel interspersed with clay layers, the distance of migration may be the greatest, because the clay layers tend to prevent the vertical movement and venting of the gases from the soil. Freezing of the ground surface and extensive rainy weather also seem to contribute to the increased lateral migration of these gases. Therefore, vegetation growth problems are found not only on former landfills, but frequently in areas adjacent to landfills.

Field Measurements

To determine whether or not foreign gases are present in the soil atmospheres, it is necessary in most cases to first make a hole in the ground. We use a commercial bar hole maker to obtain a 3 ft. deep, 1/2 in. diameter hole in the ground. This commercial instrument incorporates a steel hole-making rod and a driving weight into one convenient unit. This same type of unit is used by most gas utility companies when searching for leaks from their underground pipes. The handle of this bar hole maker is electrically insulated for safety to prevent a shock should you come in contact with a live underground electric wire.

The most convenient test to make in checking for gases of anaerobic decomposition of organic matter is for combustible gases with a combustible gas meter. A gas sample is drawn from the bar hole through an M.S.A. Explosimeter, which is one of the types of instrument used by the gas utility companies when looking for leaks in their underground lines. The Wheatstone's bridge principle is used within the instrument for determining the concentration of combustible gases. One leg of the bridge consists of catalytic unit that burns the combustible gases--changing its resistance, thereby unbalancing the bridge and giving a reading on the galvanometer. The sample is withdrawn from the bar hole by use of a 3 ft. long nonsparking probe. If desired, a nonconductive probe may be used. A rubber stopper is placed over the upper end of the sampling probe to help seal the bar hole from the ambient air. However, the nature of the sampling method frequently incorporates large quantities in dilution air. These combustible gas reading instruments indicate percent of the lower explosive limit of the gases for which the instrument is calibrated. The lower explosive limit for methane is a 5% dilution in air. However, it is possible to tell from the response of the meter whether or not the combustible gas concentration is between the lower and the upper explosive limits or above the upper explosive limit (15% methane in air). By the use of a dilution tube on the intake side of the meter, it is possible to theoretically determine the actual combustible gas concentration up to 100%.

The instruction manuals should be followed closely when using these meters and the instruments should be maintained and calibrated regularly. Calibration equipment is available, and the instruction booklet will inform you as to the frequency and extent of routine maintenance.

Another type of combustible gas testing meter, frequently used by the gas companies, operates by determining the thermal conductivity of the gas mixture. The meters are calibrated for methane in air. However, when measuring landfill gases carbon dioxide is frequently a major component of the mixture. Under these conditions the thermal conductivity meters will give inaccurate readings for the methane concentration. Therefore, we suggest that such meters not be used to measure the concentration of methane in landfill gas.

The carbon dioxide and oxygen concentrations of the ground gases obtained from the bar holes are analyzed by an Orsat method, which is normally used to measure the efficiency of fossil fuel-fired furnaces. In our field test work, we use the Bacharach Fyrite carbon dioxide and oxygen indicators. In the carbon dioxide indicator, the carbon dioxide is absorbed in a potassium hydroxide solution. In the oxygen indicator, an acidic chromous chlorine solution is used. Carbon dioxide indicators are available for reading 0 to 20% and 0 to 60% concentrations. The oxygen indicators determine 0 to 21% concentrations.

Unpleasant ground gas odors are frequently an indication of the presence of the gases of anaerobic decomposition of organic matter. These odors can be checked for by withdrawing a soil sample from the ground and smelling the sample. If the unpleasant odors of anaerobic decomposition are present, you will know it without having to receive any instructions.

Soil temperatures are recorded in degrees F by means of a thermometer with a 3 ft. long stem. The sensitive section of the stem is the bottom four inches.

Vegetation Death Problems in N.J.

Our first experiences with vegetation death due to landfill gases were in N.J., and involved the lateral migration of the gases from landfills to vegetation on adjacent properties. In these cases, the gases moved from the landfill laterally into the undisturbed ground where they occupied the root zones of the vegetation leading to the eventual demise of the vegetation. In some of the instances corrective measures were implemented; some of which were successful and some not.

In the early 70's a peach farmer in Glassboro, New Jersey experienced the death of acres of peach trees following the filling of an adjacent 20 ft. deep former sand and gravel pit with municipal domestic refuse. The pit belonged to the peach grower who had planned to have it filled with the refuse, for which he was paid by the municipality, and then he planned to extend his peach orchard over the completed refuse landfill. However, he found that the trees planted adjacent to this landfill began to die a year or two after the refuse had been placed against the bank nearest his orchard. Our examination showed that the soil in areas where the trees had died contained high concentrations of the gases of anaerobic decomposition. Landfill gas concentration gradients indicated that the gases flowed from the landfill. Although the landfill was only 20 ft. deep, the gases traveled out 70 or 80 ft. causing death and injury to the peach trees. This problem was repeated when young new peach trees were planted in these gassed soil areas.

The farmer no longer attempts to plant trees in those areas which contain the gases of anaerobic decomposition, but no control measures have been taken to prevent the lateral migration of these gases. Time will eventually solve the problem as the refuse material degrades to where it no longer produce sufficient anaerobic gases to cause problems. However, this might take scores of years.

The township of Cherry Hill, New Jersey has experienced landfill gas migration problems since 1968 when a homeowner noticed that his backyard vegetation was dying. The backyard of this home and a couple dozen others abut a former sand and gravel pit which the municipality was filling for eventual use as a park. Gases were eventually detected up to about 90 ft.

from the landfill, and in two of the homes fires were caused by these gases. Problems of dead vegetation and the entrance or potential entrance of the combustible gases into the homes ultimately involved more than a dozen and a half homeowners. These problems were taken into court. After a week of testimony, the case was settled out of court with the municipality paying about $60,000 to the homeowners and installing a convective gas-venting system around the landfill. This system has worked in part; however, there are still places where the gases are bypassing the system and moving into the backyards and perhaps even into one or two of the homes adjacent to the landfill. Fortunately, no fires have been reported within the homes since the installation of these venting systems.

The township is still attempting to transform the former landfill into a park. Part of this effort has involved the planting of scores of trees. However, most of these trees have died during the past year and a half. Our examinations indicate that, although landfill gas is probably the cause of death of a number of these trees, many others appear to have died because of poor transplanting practices. In a number of cases the roots were pruned too extensively during transplanting. In other cases there seems to have been a lack of adequate watering following transplanting.

For a number of years we have been working with a farmer in Cinnaminson, New Jersey who has experienced the death of tomatoes, corn, sweet potatoes, and rye cover crop because of migrating landfill gases. Over a period of about 5 years the distance of lateral migration of these gases gradually increased to more than 800 ft. from the nearest edge of the landfill. These gases would appear in oval patches in the farm fields where they caused the death of vegetation.

A number of systems have been tried by the landfill operator to prevent this migration. He first installed a 15 ft. deep trench filled with stone as a venting mechanism. Next, at the edge of the landfill, he installed a series of vertical venting pipes to depths of 30 to 40 feet for convective venting of the gases. However, the bottom of the landfill was reported to be 50 to 60 ft. below the farm field; therefore, these convective vents did not stop the continuous lateral extension of the landfill gases. Next, a number of soil logs were made to determine the actual depth of the landfill gases and the water table plus the character of the subsoils. These tests revealed that the soil consisted of a series of horizontal sand and gravel layers interspersed with horizontal clay beds. Apparently the clay beds kept the gases down and caused them to migrate laterally great distances. These soil logs also revealed that combustible gases were found at the 60 ft. depth, beneath a deep clay layer just above the water table. It was probably this sand strata which conducted the gases the furthest distance.

During the summer of 1976 the landfill company installed a series of deep vertical wells to evacuate the landfill gas from this deep stratum by an induced draft pump. Tests we conducted in the fall of 1976 gave a preliminary indication that this system has been successful. For the first time in a half dozen years we could find practically no combustible gas in the farm fields. A cover crop of rye has been planted, and there was no sign that landfill gases had caused damage to the crop last fall. Apparently this positive extraction system has pulled the gas out of the farm field. We hope that it will continue to operate successfully to prevent the gases from re-entering the farm field. We do not know how long it will be necessary to continue to operate this protective system which requires an electrical supply and regular maintenance.

Prior to the successful operation of the positive extraction system, there was also a problem with the potential for the gas to enter one of the farm houses. Combustible gases were detected right next to the exterior of the house foundation, and it was necessary to install an automatic monitoring and alarm system within the home to prevent a tragedy. However, fall 1976 tests indicated that these gases are no longer a hazard to the home.

Mail Survey of U.S. Landfill Vegetation Problems

In May 1975 our Rutgers group was awarded a USEPA grant for a 2 year study of problems associated with vegetating landfill sites. The first phase of the research entailed a mail survey to determine the national scope of landfill vegetation problems. Approximately 1000 questionnaires were mailed out to individuals or agencies that we felt could furnish such information. We received replies from all 50 states plus Washington, D.C. and Puerto Rico describing a total of 471 sites. No problems were reported by 356 sites, 31 reported vegetation growth problems adjacent to landfills, and 120 indicate problems with growing vegetation on landfills. Therefore, about 30% of the reported sites said they had some sort of vegetation growth problem. A number of sites were reported more than once, which accounts for the lack of agreement between the total and component numbers.

On-site Visits to Former Landfills Throughout U.S.

On the basis of the returns from the mail survey representative sites in eight major meteorological areas were selected for field observations. Various members of our research team have made field trips to most of these locations.

Field observations included the evaluation of plant species on and/or adjacent to former landfills and the selection of healthy and poorly growing specimens of a given species for an in-depth examination. Analyses for combustible gas, carbon dioxide, and oxygen were made of the soil atmospheres in the root areas of many trees and other vegetation exhibiting poor and good growth. The temperature of the soil at these locations was also measured, and soil samples were returned to the Rutgers Soils Laboratory for analysis.

Almost invariably, a high correlation resulted between high concentrations of combustible gas and/or carbon dioxide (CO_2) and/or low oxygen (O_2) and a poor growth status for the tree or shrub. Examples of this relationship are presented for the Northeastern U.S. region (cool summers and cold winters) in Table 1. At a Battle Creek, Michigan landfill a red pine exhibiting only a needle tip necrosis contained no combustible gas, 6.5% CO_2, and 19.5% oxygen in its root atmosphere at a 1-foot depth. The root zone of a dead red pine, on the other hand, contained 25% combustible gas, 21% CO_2, and only 12% oxygen. Similarly, a healthy, 20-ft. tall willows on an Auburn, N.Y. landfill contained no combustible gas or CO_2 and 20% oxygen at the 1-ft. depth; whereas beneath a nearby dead willow there was combustible gas and decreased oxygen concentration.

Results for the Southwestern California region (hot summers, warm winters, with a concentration of the year's modest precipitation in the winter) are represented by data from the South Coast Botanic Garden in Los Angeles which has been built on a former landfill (Table 2). Here, a Cytisis racemosis in good condition contained no combustible gas or CO_2 at the 1 ft. depth, whereas beneath a dead Cytisis combustible gas had risen to 22%, CO_2 to 15%, and oxygen content had decreased to 11.5%. Somewhat similar relationships are shown for Melaleuca, Aleppo pine, and Eucalyptus.

Excellent correlations were found in coastal northwest area (temperate oceanic climate) between high concentrations of combustible gases in the soil atmosphere and dead vegetation or no growth (Table 3). The two sites in Seattle, Washington are over former landfills while the Day Island site is a woodlot adjacent to a former landfill. Over the landfills it was the barren spots which had high concentrations of combustible gas near their surface while none or only a trace of combustible gas could be found in the good growth vegetation areas.

Other examples of good correlations between high concentrations of landfill gases and poor or dead vegetation are given in Table 4. Here representative comparisons are cited from Alabama (subtropical humid climate), Montana (semi-arid cold winter-steppe climate), and Long Island, New York (temperate continental-warm summer climate) areas. Sites from on and off the former landfills are presented, and each indicates that poor vegetation growth is associated with the presence of landfill gases in the soil atmosphere. However, since these field measurements are made only at one moment in time we do not always find a positive correlation. We have found that landfill gases do come and go from an area; therefore, it is possible for them to be present, do their damage and then leave. It also takes some time for the adverse effect of their presence to manifest itself in plant symptoms, so it would be possible to find the gases to be present for a short time at the base of an apparently healthy plant.

Table 5 records the "classical" field observations we might expect to find associated with the lateral migration of landfill gases and its associated vegetation death. The Holtsville Landfill is in a former sand pit, and the landfill gases have migrated through the soil into an adjacent woodland. Where the landfill gases are present in the soil the trees have died, and we find the soil in a septic condition. The temperatures of the soil containing the landfill gases are also somewhat higher than the soils without the landfill gas. Death of vegetation and associated septic soil conditions are frequently found adjacent to landfills placed in former sand and gravel mining pits.

Soil Temperature Abnormalities

Although high temperatures are not normally associated with anaerobic decomposition we frequently, but not always, find higher soil temperatures associated with the presence of landfill gases. Occasionally, we find very high soil temperatures associated with landfill gases. In January 1976 we experienced our first such high readings at the South Coast Botanic Garden in Los Angeles. Here we recorded a soil temperature of 140°F. at a 42" depth. This was in an area where the cover soil had cracked and landfill gases were flowing from the crack. All vegetation adjacent to the crack was dead. Armand Sarinana, Superintendent of the South Coast Botanic Garden, attributes many of his poor vegetation growth problems to extreme soil temperatures(6). Sarinana reports that areas of extreme soil temperatures, 120° to 160° F, are localized in the garden while more general areas of lower temperature, 70° to 90° F, are common throughout the garden. This is 15° to 20° warmer than the average home garden soil temperature in the vicinity of this facility averages between 55° and 70° F. Sarinana writes that once the soil temperatures are dropped vegetation can be re-established.

An approximate 8° F differential in temperature was found between no gas and landfill gassed soils adjacent to the Huntington Landfill (Figure 1). The higher temperatures were found in the gassed soils. A similar situation existed on and adjacent to the Day Island Landfill in Oregon where the higher temperatures were found in the soils containing landfill gases (Figure 2). Here the highest and lowest temperatures were found off the landfill where the gassed soils were more than 30° F. warmer than the ungassed soils. At Day Island, the soil temperature decreased with increasing depth while the opposite was true at Huntington.

Many reasons have been attributed to the cause of these occasional high soil temperatures including: composting, chemical reactions, and underground fires. The highest soil temperature we have recorded to date associated with a landfill was 154° F. at the 3 ft. depth in uncovered refuse at the Fountain Avenue Landfill in New York City on November 23, 1976. The area with this high temperature was "steaming" as the moisture in the gases leaving the landfill condensed.

Soil Quality Data - Gassed vs. Non-gassed Soils

Top and subsoil samples from each of six meteorological regions were analyzed for content of major and trace nutrients, pH, moisture, organic matter, conductivity and for soil texture. Table 6 contains a summary of the data expressed as percent change (+ or -) in each constituent as the soil proceeded from a no landfill gas to a high landfill gas condition.

There was very little difference in content of the major nutrient elements (magnesium, phosphorus, potassium, and calcium) between gassed and ungassed soil. Since these elements are normally present in soil in hundreds or thousands of pounds per acre, a small percentage fluctuation in content would have a negligible effect on plant growth.

Nitrogen compounds (NO_3-N and NH_4-N) and trace elements (iron, manganese, zinc, copper, and boron) which are normally present in much lesser quantity, increased many fold in soils with high concentrations of landfill gas in their atmospheres. In particular, the ratio of iron to manganese, a critical value in soil fertility, was frequently above the recommended range for adequate plant growth.

Conductivity, which is a measure of total ion activity, was, understandably, increased as well.

Soil pH was either increased or decreased depending on the original condition of the soil; alkaline soils such as those in Utah and Idaho decreasing, while the more acid soils of the Northeast increased in pH value.

The reason for the observed increases in content of nitrogen compounds and trace elements in gassed soils undoubtedly lies in the low redox potential of these soils as has been documented for similar responses of soil to flooding conditions. When oxygen disappears from the soil, requirements of anaerobic soil microorganisms for a source of oxygen results in the reduction of several oxidized compounds namely nitrate, nitrite, and the higher oxides of manganese, and iron. These reduced forms are generally more soluble and hence are made available to plants. Availability of other trace metals occurs as they are displaced by ferrous ions from the exhange complex to the soil solution.

The trend to neutrality in pH is caused by the buffering effect of organic acids released by the microbial breakdown of organic matter.

The consequences of these soil changes in landfills have yet to be evaluated. However, it is not improbable that trace element toxicities may also play a role in the detrimental effects of landfill conditions on plant growth.

Field Evaluation of Possible Anaerobic Soil Gas Vegetation Problems

Certain basic information is valuable for use in the field to determine whether or not the potential exists for a vegetation viability problem caused by gases from buried organic refuse. It is always advisable to ask about the property to determine whether or not it had ever been a refuse landfill or if there had been such an operation adjacent to the property under consideration. However, information obtained by word of mouth may not always be reliable. Therefore, the following criteria are suggested for on-site evaluation of the potential for a landfill vegetation growth problem:

1. Is there unhealthy or dead vegetation in the vicinity?

   Obviously, many things could cause poor quality or dead vegetation, but there is always the possibility that it might have been caused by landfill gases.

2. Do you notice any unpleasant odors in the vicinity?

   Again, unpleasant odors come from many sources. Those odors from landfills or soils turned septic by landfill gases are generally of a putrid nature. In moving over a former landfill you can occasionally detect them in the air as they are discharged through surface cracks, or as you work with the soil you may notice the septic odor.

3. Examine the soil. It is easy to draw a soil sample and examine it in the field.

Table 7 may be used as a general guide to soil aerobic/anaerobic conditions. Under certain circumstances there will be exceptions to these general guidelines. However, those who work with soils know the typical characteristics of a healthy soil and will easily recognize the qualities associated with an anaerobic soil. Soil temperature differences are sometimes extreme and sometimes very modest. We have recorded temperature differentials as high as 30° F. between a normal healthy aerobic soil and a nearby anaerobic soil. However, in most cases the difference is very small, and occasionally we find the anaerobic soil at a lower temperature than the aerobic.

The farmer in Cinnaminson, N.J. felt that something was wrong with his soil for two reasons. One was the septic odor released when he plowed it, and secondly, the unnatural way in which the furrow lay during plowing. Instead of the soil falling in an even mound, it fell in clumps; the soil did not break apart easily as is natural for this soil.

The higher moisture content of anaerobic soils is possibly due to the condensation of the water vapor generated by the decomposition of the organic matter within the landfill. The darker color of these anaerobic soils is probably due to their septic nature.

Of course, one can always look for landfill gases in the soil by means of the instrumentation previously described.

Finally, the area should be observed for indications of any unusual settlement. Landfills left unused for a while develop uneven settlement. This plus a number of surface cracks in the soil are sometimes indicators that there might possibly have been organic refuse buried here in the past.

Control Measures to Reduce Vegetation Losses

Measures for control of landfill vegetation growth problems involve: construction of the landfill; vegetation cultural methods including proper fertilization, soil amendments, and irrigation; removing the landfill gas from the vegetation area; the use of tolerant plant species; and suitable planting techniques. These control techniques are summarized in Table 8.

Research is now being conducted at Rutgers in order to investigate some of these factors. One project involves the screening of 19 woody species for tolerance to landfill growth conditions.

The experimental plot is located on a former landfill in New Brunswick, New Jersey. A nearby natural woodland site was cleared as a control area. The 19 species were planted in replicates of 10 and observations are being made for tree height and girth, shoot elongation, fruiting, and the presence of heart rot. The rhizosphere of each tree is also being analyzed for gas composition, including combustible gas, carbon dioxide and oxygen, and for soil temperature. Soil samples are also taken for analyses of the chemical constituents of the soil.

Besides the screening program, five planting techniques have been employed, including two mounds and three trenches with no gas protection or underlain either by 1 ft. of clay or by a plastic sheet over a gravel base, with and without vertical venting pipes.

A second experiment is being carried out in the greenhouse, and entails an investigation of the effect of varied concentration ratios of methane; carbon dioxide; and oxygen circulated through the root zones of seedlings growing in sand/solution culture.

Preliminary results appear to indicate that tomato plants whose roots were exposed to a gas mixture containing methane, carbon dioxide and oxygen roughly in the proportions in which they exist in anaerobically respiring landfills, suffered growth alterations which were more marked than those sustained by similar plants which were merely deprived of a normal oxygen supply.

Meaningful results of the field and greenhouse experiments will not be immediately forthcoming. However, when the data have all been collected it is hoped that we will have a better understanding of landfill vegetation problems and that we will be able to make recommendations of species and planting techniques for greater success in the vegetation of former landfill sites.

Conclusion

The field examinations we have conducted of landfill vegetation throughout the United States indicate that poor growth or death of the vegetation is directly associated with the presence of landfill gases in the soil atmosphere. It appears that the vegetation dies as the soils become anaerobic from the physical displacement and/or biological consumption of the soil oxygen.

To obtain a successful growth of vegetation above or adjacent to former refuse landfills the root zone of the vegetation must be protected from the gases of anaerobic decomposition of the organic matter in the landfills.

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TABLE 1

FIELD DATA N.E. (1975)

Site	Species	Condition	0 at 1' Depth		
			Comb. Gas	CO_2	O_2
			Approx. Volume %		
Auburn, N.Y.	Willows	20' Tall	0	0	20
	Willows	Dead	15>5	0	18.5
Battle Creek, Mich.	Red Pine	Tip Necrosis	0	6.5	19.5
	Red Pine	Dead	25	21	12
Guilderland, N.Y.	Poplars	4'-8' Tall	10	15	19
	All Veg.	Dead	>50	36	2

Table 2

Los Angeles Area,

Field Data Cal. (1975)

Site	Species	Condition	Volume % at 1' Depth		
			Comb.	CO_2	O_2
S. C. Bot. G.	Cytisis Racemosis	Healthy	0	0	19.5
	Cytisis Racemosis	Dead	22	15	11.5
S. C. County Pk.	Melaleuca	Healthy 12'	4	5	7
	Melaleuca	Dead	>50	43	3.5
S. C. County Pk.	Aleppo Pine	Green 65"	<1	0.5	17
	Aleppo Pine	Chlorotic	>50	25	11
M. G. Golf Course	A. Pine & Eucalyptus	Good	0	0	20
	A. Pine & Eucalyptus	Poor	>50	5	9.5
Miss. Canyon	A. Pine	Healthy	0	0	21
	A. Pine	Yellow Color	0	1	19.5
	A. Pine	Yellow Tips	<1	6	20

Table 3

N.W. U.S. Area

Field Data (1976)

Site	Species	Condition	Approx. % Combustible Gas at Various Soil Depths		
			1'	2'	3'
University of Washington Seattle, Wash.	2' Tall Alfalfa Clover, Rye, and Vetch	Healthy	0	0	0
(On Former Landfill) " " "	None	Barren Ground	15>5	15>5	20
Genesee St. Park Devel. Seattle, Wash.	Grass and Weeds	Healthy	0	0	—
(On Former Landfill) " " "	None	Barren Ground	>15	>15	—
Day Island Landfill Lane County, Oregon	White Ash and Broad Leaf Maple	Healthy	0	—	0
(Adjacent to Landfill) " " "	" " "	Dead	15>5	—	>50

TABLE 4

FIELD DATA (1976)

Site	Species	Condition	Approx. % Combustible Gas at Various Soil Depths		
			1'	2'	3'
Old Dothan City Landfill Ashfor, Alabama	Loblolly Pine (25' tall)	Healthy	0	0	0
(Adjacent to Landfill) " " "	Loblolly Pine (20' tall)	Dead	5	>50	>50
Great Falls San. Landfill Great Falls, Montana	Wheat	Good Growth	0	T	20
(on Former Landfill) " " "	Wheat	Very Poor	28	>50	---
Kings Park, San. Landfill Smithtown, L.I., N.Y.	White Oak (30' tall)	Living	T	---	---
(Adjacent to Landfill) " " "	White Oak (30' tall)	Dead	>50	---	---
" " "	Hemlock (7' tall)	Living	0	---	---
" " "	Hemlock (6' tall)	Dead	8	---	---

TABLE 5

SOIL AND VEGETATION CONDITIONS ADJACENT TO
HOLTSVILLE LANDFILL
BROOKHAVEN TOWNSHIP, L.I., N.Y.
10/14/76

Approx % Combustible Gas at Various Soil Depths			Soil Conditions	Species	Conditions	Ground Temps (^{o}F)
1'	2'	3'				
30	>50	>50	Septic Odor Dark Color Damp	Red Oaks	Dead	70 - 80
0	0	0	Normal Soil Odor	White Oaks	Live	62 - 67

TABLE 6

PERCENT (%) CHANGE IN CONTENT OF CONSTITUENTS AS SOIL PROCEEDED FROM NON-LANDFILL GAS TO HIGH-LANDFILL GAS CONCENTRATIONS

Soil Constituent	Meteorological Region Dca N.J.	 Dcb N.E.	 Do N.W.	 Cf Ala.	 Bs Utah	 H Idaho	 Mean
Lb/Acre							
Mg	- 5.6	- 6.2	± 0	- 9.2	± 0	+ 1.6	- 3.2
P	- 11.1	- 13.6	+ 13.9	- 4.8	+50.0	+37.5	+ 11.8
K	+ 27.3	+ 33.9	- 26.0	-20.4	± 0	+ 9.0	+ 4.0
Ca	- 17.1	+ 19.5	+ 3.8	+50.9	+ 8.3	-13.4	+ 8.7
NO_3-N	+ 170.2	+ 2.2	- 7.6	-17.5	-15.0	+23.9	+ 26.0
NH_4-N	+ 93.8	+ 222.0	+800.0	-47.1	-58.3	-52.6	+159.6
%							
H_2O	+ 37.8	+ 6.8	- 12.7	- 7.5	+14.3	+ 5.3	+ 7.3
Org. Matter	- 50.0	- 35.2	- 9.3	+ 9.4	-22.1	+79.1	- 4.7
Sand	- 4.4	+ 3.8	- 12.0	+14.4	+10.0	+12.6	+ 4.1
Silt	+ 100.0	- 13.7	+ 8.8	-31.1	- 5.9	-17.3	- 6.8
Clay	+ 11.1	- 9.3	+ 10.5	-26.7	-23.0	- 7.5	- 7.5
ppm							
Fe	+ 146.8	+15,500	+ 39.6	+28.3	-	+28.0	+3,148
Mn	+ 65.4	+ 125.0	+ 25.8	+19.0	-	+286.0	+104.2
Cu	+ 10.8	+ 200.0	+ 38.3	+70.5	-	- 28.0	+ 58.3
Zn	- 6.7	+ 785.0	+ 63.5	+410.0	-	+370.0	+324.4
B	+ 2.4	-	+ 67.5	- 3.7	+ 2.6	+ 3.4	+ 14.4
Fe/Mn	+ 49.6	+7150.0	+ 10.8	+ 7.8	-	- 68.6	+1430.0
Cond. (Mmohs)	+ 210.0	+ 66.7	+117.0	+210.0	± 0	+100.0	+ 117.3
pH	- 5.6	+ 3.5	+ 7.8	+ 7.2	- 1.2	- 2.4	+ 1.6

TABLE 7

GUIDE FOR EVALUATION OF LANDFILL SOIL GAS PROBLEM

Characteristic	Anaerobic Soil	Aerobic Healthy Soil
Odor	Septic	Pleasant
Color	Darker	Lighter
Moisture Content	Higher	Lower
Frability	Poor	Good
Temperature	Higher	Lower
Combustible Gas	Higher	Very Low or Zero
Oxygen	Lower	Higher
Carbon Dioxide	Higher	Lower

TABLE 8

CONTROL TECHNIQUES TO REDUCE VEGETATION LOSS

1. Suitable Species -
   Shallow Rooted, Adapted to Anaerobic Conditions
2. Cultural Methods -
   Adequate Lime, Fertilizer, Top Soil for Cover, Irrigation
3. Soil Amendments -
   Shredded Refuse, Mulch, Sewage Sludge
4. Landfill Construction -
   Proper Grading, Compaction, Adequate Depth and Quality of Cover Soil and Top Soil
5. Planting Techniques -
   Vented or Lined Trenches, Mounds, or Gas Barriers
6. Gas Removal by Induced Draft Evacuation

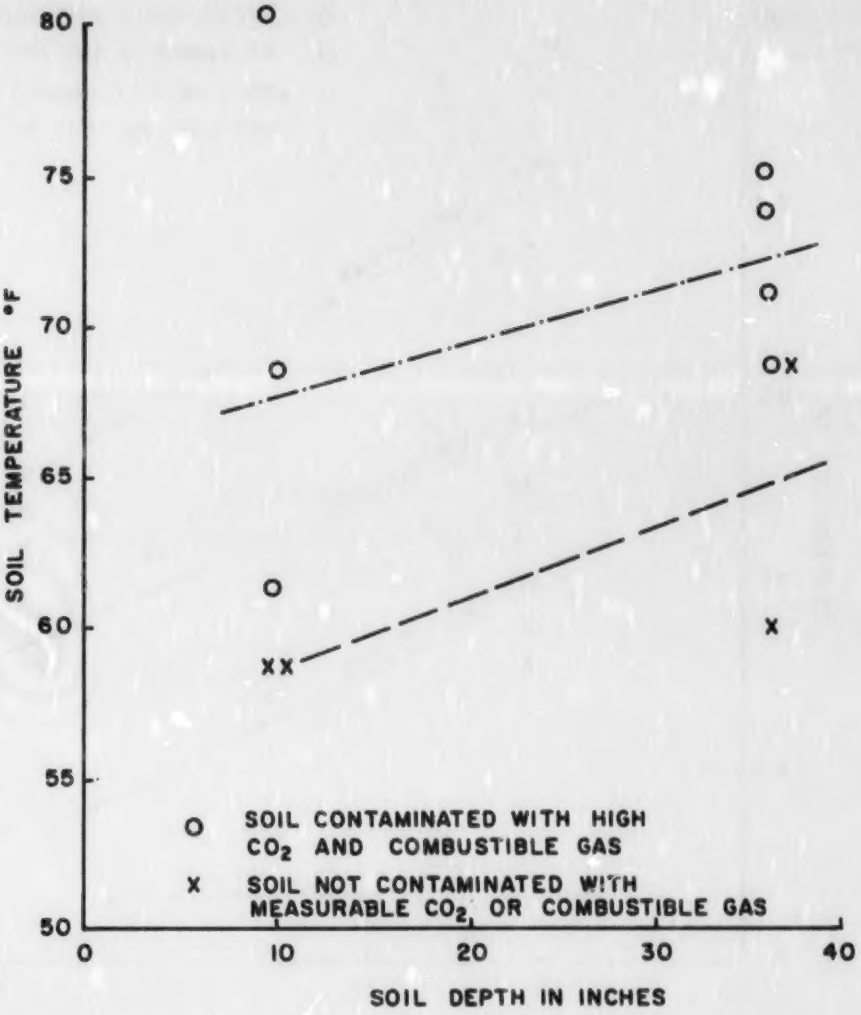

Figure 1

SOIL TEMPERATURES, IN THE VICINITY OF HUNTINGTON LANDFILL, HUNTINGTON, L. I., N.Y.
10/15/76

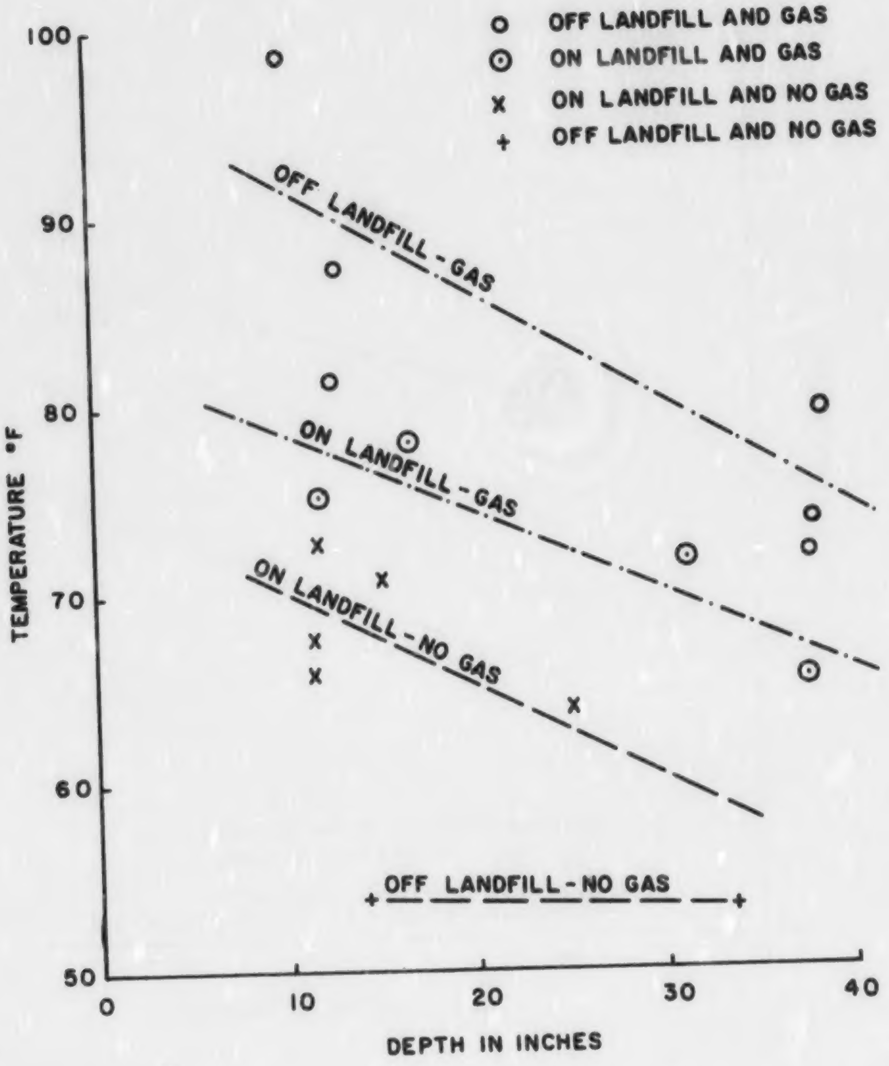

FIGURE 2

SOIL TEMPERATURES, DAY ISLAND LANDFILL
June 24, 1976

EFFECTS ON SOILS AND PLANTS FROM APPLICATIONS OF COMPOSTED MUNICIPAL SOLID WASTE - A SUMMARY OF SELECTED RESEARCH PROJECTS

by

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ABSTRACT

This paper presents a summary of results from research projects conducted to help determine the effects on selected soils and plants from applications of composted municipal wastes. The studies included investigations in greenhouses, field investigations, and limited demonstrations conducted from 1969 through 1975. Positive and negative responses to the waste applications were observed which related to increased and decreased crop yields, soil improvements, and increases in some heavy metals in plant tissue.

INTRODUCTION

Composting has had a less than satisfactory history in the United States. Although technically feasible, poor market and economic factors have resulted in the failure of composting to be widely practiced as a solid waste management process. Most composting studies now being conducted are aimed at providing an acceptable disposal option for the high amounts of sewage sludge requiring disposal. In earlier studies, EPA evaluated composting as a method for managing United States municipal solid waste, including sewage sludge. Among these evaluations were the research and demonstration at Johnson City, Tennessee (open windrow) and Gainesville, Florida (mechanical high rate digestion).

Although these composting projects included provisions to evaluate the utilization of the composted product, major investigations concerned the evaluation of the process, equipment performance, and economics. Additionally, because of changes in the overall solid waste R&D mission and redirection of available resources to higher priority needs, the projects were terminated before meaningful marketing studies and utilization evaluations could be fully implemented. There were, however, some efforts to demonstrate the benefits of using the compost for reclaiming poor soils and increasing crop yields. Dr. Hortenstein at the University of Florida utilized material produced at the Gainesville plant. Results of these studies have been reported in a number of publications.* Compost utilization efforts at Johnson City, Tennessee, initially involved applications of compost to strip mined areas for reclamation, and limited applications to crop producing lands. Responses were measured by recording visual observations and by crop yield information. Prior to the termination of the Johnson City project, plans were formulated to increase the utilization studies and to include greenhouse as well as field evaluations of the effects of the composted wastes when applied on selected soils and plants. One of the driving factors to implement these studies was a concern about potentially toxic materials such as heavy metals which might be carried in the compost and transmitted to soils and further to ground water and food crops. A number of projects were initiated prior to the plant's closing and provisions were made to permit selected studies to continue at reduced levels after the Johnson City composting plant closed. Those studies continued included projects in the vicinity of Johnson City, Tennessee, at the National Fertilizer Research Center at Muscle Shoals, Alabama, and at other miscellaneous locations.

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The purpose of this paper is to describe selected portions of the Johnson City compost utilization research and demonstrations and to summarize major results. The studies were conducted during the period 1968 through 1975. Details are available.*

INVESTIGATIONS

For purposes of organization, the investigations are divided into three categories, i.e., Field Studies, Green House Studies, and Others. Field studies involved application of varying amounts of composted municipal solid waste-sludge mixtures, sewage sludge and chemicals to selected crops under field conditions. Greenhouse potting experiments were conducted under more controlled conditions and in some cases under accelerated time schedules. The other category includes the corn experiments at the Johnson City location, demonstrations, and other miscellaneous projects which primarily involved land reclamation.

Field Studies

Field studies at Muscle Shoals, Alabama, included evaluation of the responses of forage sorghum and common bermuda grass to Municipal Solid Waste (MSW) compost; responses of sweet corn and string beans to zinc and other heavy metals contained in municipal wastes; measurements of heavy metal content of several vegetable species grown in soil amended with sewage sludge, and others.

Sorghum and Bermuda Grass

These experiments were among the first to be implemented and were designed primarily to determine the crop growth responses to compost and to determine compost application rates which would be detrimental. In the first experiment, forage sorghum was the test crop. Thirteen treatments comparing combinations of fall and spring applications with and without supplemental N fertilizers were compared. Compost at total application rates ranging from 23 to 326 metric tons/hectare (ha) (dry-weight basis) were applied over a 2-year period. All plots were fertilized with P and K at rates determined necessary from soil analysis according to acceptable practice. All treatments were replicated four times.

The sorghum forage was removed in two cuttings each year from 1969 through 1971; but was cut only once in subsequent years. In addition to measuring crop production, forage samples were analyzed for several macro- and micronutrients.

In the 1968 experiments involving common bermuda grass, compost was applied as a top dressing at rates of 0, 9, 18, and 27 metric tons/ha. One year later, compost was again applied, but at triple the 1968 application rates. Nitrogen was applied to these plots after the second compost applications. The grass was harvested four times.

Summary Results on Sorghum and Bermuda Grass

Higher yields were observed with annual compost applications of rates up to 143 and 80 metric tons/hectare (ha) on sorghum and bermuda grass, respectively. However, the highest yields of either crop attained with compost were surpassed by application of fertilizer nitrogen (N) at the rate of 180 Kg/ha with adequate phosphorus (P) and potassium (K). In addition to increasing the fertility of the soil, compost applications increased moisture-holding capacity and decreased the bulk density and compression strength of the soil. These effects were observed for three years. Although the results showed that relatively large tonnages of compost can be applied to grass or crop land, the measured economic return in terms of increased crop yields versus application expenses was minimal. Of further concern was the observed elevation of Zn and to a lesser extent other heavy metals. These were largely derived from the sewage sludge in the composted refuse. Additional details of these experimental results are provided in Tables 1 through 5 and in Figure 1.

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*A Bibliography of information available from this work is included.

Availability of Zn in Compost and Sewage Sludge

The second series of investigations were initiated to compare the availability to corn and string beans of Zn in compost and sewage sludge with comparable rates of $ZnSO_4$. These studies were prompted by the elevated levels of Zn found in the forage sorghum grown on compost treated soils.

Appropriate species of sweet corn and string beans were selected as the test crop because of their contrasting response to high levels of available Zn. Soil treatments consisted of three rates each of dry garbage compost (56, 112, and 224 metric tons/ha), dry sewage sludge (50, 100, and 200 metric tons/ha), and $ZnSO_4$ all incorporated into the soil to provide the equivalent of 90, 180, and 360 Kg of Zn/ha. Treatments were replicated 4 times. The crop area was irrigated as required to assure that moisture stress would not limit plant growth.

Sweet corn was planted so that there were 4 rows in each test plot and later thinned to a population of 50,000 plants/ha. The corn was harvested in the milk stage. String beans including vines were harvested from two center rows of each string bean plot. Soil borings were also taken from all plots shortly after planting corn each year. These were analysed for Zn, Cu, Pb, and Ni by atomic absorption spectroscopy and Cd and Cr using a flameless atomizer accessory. At harvest the two center rows of each plot were cut, weighed, and total dry matter production and yield of the edible portion were measured. Plant material samples were analyzed for Zn, Cu, Cd, Cr, Pb, and Ni.

Summary of Results from Zn Availability Experiments

The experiments were initiated in the fall of 1971 and continued until 1975. The overall results indicated that when $ZnSO_4$ and compost or sewage sludge were applied at rates resulting in comparable applica-

TABLE 1. CHEMICAL ANALYSES OF COMPOSTS USED (%, DRY WEIGHT BASIS).

Application time	N	P	K	C	Ca	Na	Mg	S	Zn
Fall 1968	1.2	0.2	0.8	30.2	3.6	0.8	0.5	0.4	0.1
Spring 1969	1.3	0.4	1.0	28.8	6.4	0.8	0.9	0.4	0.1
Fall 1969 and Spring 1970	1.3	0.3	1.0	27.3	4.6	0.7	0.6	0.5	0.1

TABLE 2. COMPOST EFFECTS ON NUTRIENT CONCENTRATION IN SORGHUM FORAGE

Compost rate, metric tons/ha								
1969	1970	Total	1970	1971	1972	1973	1974	1975
			N, %					
0	0	0	1.7	1.1	1.1	1.0	0.9	
28	18	46	1.7	1.3	1.3	1.1	1.1	
46	36	82	1.8	1.5	2.1	1.2	1.3	
91	72	163	2.0	1.7	1.7	1.3	1.3	
183	143	326	2.1	2.0	2.0	1.5	1.5	
			Na, ppm					
0	0	0	91	82	36	75	88	
28	18	46	30	88	40	75	112	
46	36	82	30	96	90	100	93	
91	72	163	64	104	46	100	91	
183	143	326	94	106	36	125	118	
			Zn, ppm					
0	0	0	32	22	19	20	28	24
28	18	46	34	32	23	26	30	29
46	36	82	41	40	45	27	34	34
91	72	163	50	51	36	27	33	33
183	143	326	57	60	40	32	40	[illegible]
			Cu, ppm					
0	0	0	4.0	3.5	3.4	3.0	5.4	3.8
28	18	46	4.4	4.5	3.6	3.0	6.2	4.0
46	36	82	4.4	5.0	5.8	3.0	6.4	4.0
91	72	163	5.1	6.8	4.8	4.0	6.7	4.2
183	143	326	5.6	7.0	5.1	3.0	7.8	5.1

TABLE 3. COMPOST EFFECTS ON SOME CHEMICAL CHARACTERISTICS OF SOIL

Total compost application metric tons/ha	Organic matter %		pH		Extractable nutrients, kg/ha K		Ca		Mg		Zn	
	1970	1973	1970	1973	1970	1973	1970	1973	1970	1973	1970	1973
0	1.6	1.2	5.4	5.6	183	109	1632	2688	181	171	7	27
46	1.8	1.4	5.7	6.4	197	100	2340	3680	190	214	29	43
163	2.7	1.6	5.6	6.8	275	135	3349	4184	224	233	91	71
326	4.2	2.0	5.8	6.9	332	168	3920	4764	234	256	400	118

TABLE 4. COMPOST EFFECTS ON SOME PHYSICAL CHARACTERISTICS OF SOIL

Total compost application metric tons/ha	Soil moisture, %		Bulk density		Unconfined compression strength, kg/cm²	
	1970	1973	1970	1973	1970	1973
0	12.4	10.4	1.37a*	1.50 n.s.	2.9a	2.5 n.s.
46	12.5	10.3	1.30a	1.45	2.9a	2.3
163	13.3	10.7	1.20b	1.44	2.4b	2.1
326	14.8	11.2	1.18c	1.43	1.9c	2.2

*Value followed by the same letter are not significantly different according to Duncan's Multiple Range Test (5%).

TABLE 5. COMMON BERMUDA GRASS YIELD, AS AFFECTED BY COMPOST AND N

Compost rate metric tons/ha			Dry forage yield metric tons/ha*		
1969	1970	N rate, kg/ha	1969	1970	Avg.
0	0	0	6.5c	2.7d	4.5d
0	0	180	11.4a	5.6b	8.5b
9	27	0	7.4bc	4.0c	5.6c
9	27	180	11.2a	6.5ab	8.7b
18	54	0	7.6bc	6.3c	5.8c
18	54	180	12.5a	7.2a	9.9a
27	81	0	7.8b	4.7c	6.3c
27	81	180	11.9a	13.4a	9.2ab

*Values followed by the same letter are not significantly different according to Duncan's Multiple Range Test (5%).

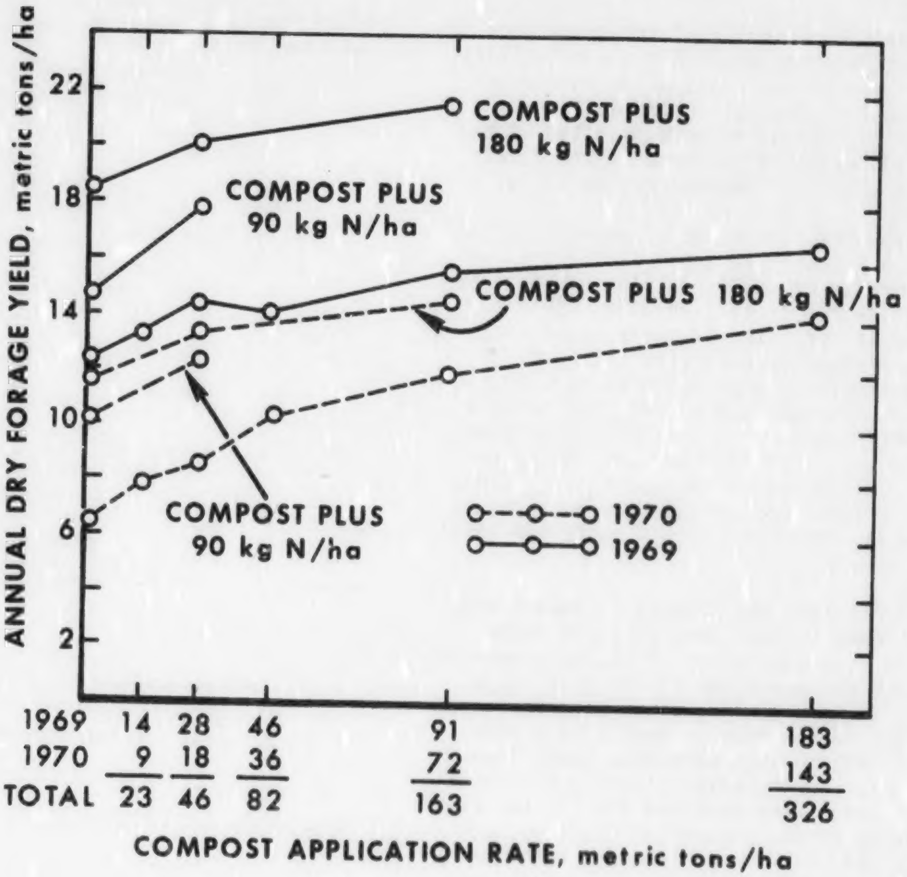

Figure 1. Effects of Compost Application on Forage Sorghum Yield.

tions of Zn, availability of Zn to both the sweet corn and the string beans was considerably greater from $ZnSO_4$ over the 4-year period. In general the string beans were less tolerant of high Zn treatments even when applied as sewage sludge. Of the other heavy metals monitored in these crops, only Cd levels exceeded check values in the plant leaves. The edible plant parts were relatively unaffected.

Bean pods, however, did accumulate Ni from sewage sludge, at levels two to four times those from nontreated plots. Although there was no indication that the elevated levels of Cd or Ni were phytotoxic, the consequence of animal or human consumption was not determined. Tables 6 through 11 are examples of results available from these experiments.

Heavy Metals in Several Species of Vegetables Grown in Soil Amended with Sewage Sludge

Additional experiments were started to compare responses of several other vegetables to heavy metals contained in two different sewage sludges. These experiments were initiated because the earlier studies indicated that corn and beans differ in capacity to accumulate Zn and other heavy metals from soils amended with municipal wastes.

These experiments were conducted in a manner similar to those already described. Anaerobically digested sludges from two different sources (Table 12) were applied at rates of 112 metric tons/ha (dry) in the fall of 1973. Seeds of all vegetables (Tables 13 and 14) were sown except peppers, tomatoes, and lettuce, which were transplanted. Adequate soil moisture was maintained as required. Vegetables were harvested from triplicate plots as they matured and total yields were measured. Samples of leaves and edible parts, in addition to soil samples were taken.

Results

Increased yields of tomatoes and squash were measured in response to applications of both the sludges; other species were unaffected. Leafy vegetables, such as lettuce and spinach, accumulated Zn, Cd, and Cu. Radish and turnip roots contained somewhat lower levels of Zn and Cd, while vegetables consumed as fruits or seed pods were relatively low in heavy metal content. Although there were significant differences in metal content between the two sludges, uptake of metals by the plants was not always related to metal content in the material. This suggests that the chemical form may be important in evaluating sludges for use on agricultural land.

Greenhouse Pot Studies

Greenhouse studies conducted during this program involved investigations dealing with the effects of soil pH on heavy metal uptake by plants grown in soil amended with sewage sludge; responses of corn to Zn and Cr in municipal wastes applied to soil; and relationship of Zn and Cd supply to uptake by fescue.

The first experiment was designed to determine whether liming an acid soil would significantly depress uptake of heavy metals present in anaerobically digested sewage sludge. Mountview silt loam was limed to pH values of 5.0, 6.0, 7.0 with a 4:1 mixture of $CaCO_3$ and $MgCO_3$. Mustard seeds were sown and thinned to 12 plants per pot and harvested. Leaves (3 times in 10 weeks) were analyzed for heavy metal content.

The second experiment was conducted to determine if Zn and Cr in several municipal waste products becomes available to plants after soil incubation and successive cropping. Experimental procedures involved the liming of the soil to pH 5.5, addition of appropriate supplemental nutrients to pots containing 3Kg of soil. Any nutrients contained in the sewage sludge or compost were in addition to the supplemental nutrients. Various cropping sequences were used to determine effects of varying the length of time between waste application and planting, planting after moist incubation (faster decomposition of the waste) and other factors.

A third experiment was conducted to determine whether varying the ratio of Zn to Cd in the soil influences their mutual uptake. This was based on the premise that excess Zn prevents toxic accumulation of Cd by first causing plant injury. General procedures used were to fertilize with NH_4NO_3, superphosphate, K_2SO_4 in amounts to provide the experimental amounts of N, P, and K desired; adding Zn and Cd as chlorides in selected quantities; and planting Kentucky 31 fescue. Clippings from each treatment were harvested monthly for 4 months, oven-dried and analyzed for Zn and Cd content.

Summary Results from Greenhouse Pot Experiments

Abatement of Cu, Pb, and Ni in mustard was not accomplished by liming the soil from pH 5.0 to 7.0. However, concentrations of Zn decreased significantly over this range and Cd decreased slightly from ph 5.0 to 6.0. Other studies are needed to evaluate liming as a means of heavy metal pollution abatement.

The pot experiments to study phytotoxic levels of chromium (Cr), Cd, and Zn indicated that inorganic forms of these metals are taken up by plants in greater quantities than from organic wastes. Whereas, Zn concentrations in corn increased with the rate of Zn applied as either $ZnSO_4$ or in organic waste, concentrations of Cr seldom exceeded check values in the plant foliage, even though growth was depressed. Apparently, Cr is effectively excluded by plants with very high soil concentrations, resulting in sufficient root damage to limit growth. Cadmium was readily accumulated in fescue, but growth was not depressed until concentrations exceeded approximately 100 parts per million (ppm). Details of these greenhouse pot experiments, experimental procedures, soil and plant characteristics, and results have been reported and are available.

Soil Mobility Studies

Limited studies were conducted to determine the potential for contamination of ground waters with heavy metals from land disposal of large quantities of sewage sludge. These experiments involved the use of leaching columns packed with combinations of different soils, appropriate treatments of nutrients, selected plants, and other factors designed to:

- determine nitrogen (N) effects on mobility and uptake by plants of heavy metals in sewage sludge applied to the columns, and
- determine the movement in soils of heavy metals from municipal wastes and inorganic sources.

Results from these limited experiments generally indicated that mobility of heavy metals is slightly greater from inorganic than from organic forms but is minimal even under severe leaching situations. Based upon these findings, it is unlikely that controlled disposal rates of municipal wastes applied to cropland under similar conditions pose major threats to our water supplies. However, the consequence of prolonged use on cropland and subsequent soil management practices is not understood.

Other Experiments

Other composted waste utilization experiments conducted as a result of the Johnson City composting project involved:

- determination of the physical and chemical effects of municipal compost containing sewage sludge on soil and corn plants at the Johnson City site,
- demonstrations involving the use of composted municipal solid waste (with sewage sludge) on selected crops, and
- demonstrations of using composted municipal solid waste for reclaiming poor soils and areas such as strip mines and road banks.

The corn experiments and the demonstrations were conducted from about 1968 and some continued after the plant closed in 1971. This paper will not attempt to discuss all the demonstrations which were conducted during the project. Selected ones will, however, be summarized to indicate the type of results obtained. The experiment with the corn will also be summarized. Additionally, summaries are included of miscellaneous projects conducted at the Mountain Horticultural Crops Research Center, Fletcher, North Carolina, and at the USDA Tobacco Experiment Station, Greenville, Tennessee.

The Johnson City Corn Experiment

The Johnson City corn experiment was conducted from 1969 through 1975 on a loam soil and was statistically divided into four (4) major groups comprised of 13 subplots. The subplots were statistically treated with various applications rates combining compost and nitrogen fertilizer. See Tables 15 and 16. Analyses completed consisted of crop yields, soil and plant tissue analyses for nutrient content. Soil samples were analyzed for pH, moisture holding capacity, moisture content, bulk density, content of organic matter, phosphorus, potassium, calcium, magnesium, zinc, nitrogen, sodium, manganese, and copper. Plant tissue was analyzed for N, phosphorus, K, calcium, magnesium, Na, Zinc, manganese, and copper.

In 1973, corn grain, cobs, and leaf tissue were analyzed for lead, cadmium, choromium, nickel, zinc, and copper. During 1974 and 1975, selected subplots were also monitored for plant yield, uptake of heavy metals, soil bulk density, and soil moisture; however, only a chemical fertilizer was applied to these subplots after 1973 (i.e., no compost was added).

The following conclusions were reached based upon the data obtained during the duration of this research. Detailed results are available.

The application of nitrogen fertilizer alone at a rate of 160 pounds per acre had an adverse effect on corn yield. This can be partially attributed to the lower pH values of the soil in subplots that received nitrogen at high rates of application.

The time of compost application at lower rates had no significant effect on corn yields; however, at higher rates, the time of application became more important. This was evident after the initial application when nutrient deficiencies were noted on plots that had received compost at a rate of 200 tons per acre. The highest compost rate also caused some germination problems in 1969 because the bulkiness of the compost prevented the formation of a firm seedbed.

There was a rather large increase in zinc accumulation in the soil and tissue of plants on subplots that received compost at high rates of application. Although the corn in this project showed no adverse effects from the zinc, other plants with less tolerance could be affected.

High rates of compost application had no significant effect on uptake of nitrogen in corn tissue. Although the compost had a relatively low nitrogen content (less than 2 percent), high application rates resulted in excessive accumulations of nitrogen in the soil. Since the high application rates did not increase the levels of nitrogen in the plant, it is conceivable that some of the nitrogen remained in the soil or was leached from the root zone. In either case, the results of high rates of compost application may prove to have an adverse environmental effect. It was beyond the scope of this research to determine the fate of the excess nitrogen.

The uptake of potassium was increased at higher rates of compost application. This increased uptake indicates that although the potassium content of compost was low, it was readily available to the corn plants. There were no significant differences in element uptake among other treatments in this project.

These data indicated that compost had a definite liming effect. This could be valuable in soils with a low pH or in those that received high rates of nitrogen fertilizer. The maintenance of nearly neutral soil conditions on the plots that received high rates of compost also helped reduce the uptake of heavy metals such as zinc. If this soil should later become acidic, then heavy-metal toxicity could become a potential problem.

The use of compost had a positive effect on bulk density throughout the project. By reducing the bulk density of compacted soils, the compost provided for better penetration of air and water and thus enhanced the development of the root system of plants grown in this soil and reduced the energy required for tillage.

The application of compost increased the level of organic matter in the soil; the addition of nitrogen fertilizer decreased this level. When both compost and nitrogen fertilizer were applied to the soil, there was no significant change in the organic matter. These results show the importance of adding compost to the soil to improve or maintain soil fertility.

The soil moisture content was increased by the application of compost. Moisture in the soil is important as a solvent for plant nutrients to enable plants to withstand extended periods of drought. This finding could become the most important result of this research as we search for improved methods of conserving moisture to enhance the production of food on arid lands.

Trends in the accumulation of heavy

metals in selected corn plant tissue revealed that metal accumulation was more pronounced in leaf tissue than in either grain or cob tissue when compost was applied at rates up to 200 tons per acre. Plants grown in control subplots had no significant differences (except in level concentrations) in the accumulation of heavy metals in different plant parts. Residual accumulation of heavy metals in young corn plants in 1974 and 1975 showed a reduction in uptake of metals. This decrease was partially due to the absence of compost applications (the source of heavy metals) during these two growing seasons. However, data show that concentrations of heavy metals persisted in the top 6-inch layer of plowed soil.

Residual corn crops, soil bulk density, and soil moisture content continued to respond favorably on subplots that had received compost applications at rates up to 200 tons per acre. Although heavy metals persisted in the soil, their effect on corn yields appeared to be minimal. Unpublished data indicate no significant differences in the content of heavy metals in corn grain in residual crops grown in 1974 and 1975.

Tobacco Experiment at Greenville Tennessee

Since tobacco is a relatively high value crop in terms of monetary return, the successful use of compost in tobacco crops might increase the compost's value. This miscellaneous experiment was conducted to determine whether any pathogens affecting tobacco were present in the compost and whether compost could replace shagnum peat moss in a normal greenhouse potting mixture.

Thirty day old seedlings of burley tobacco were transplanted to pots containing the regular potting mixture, mixture in which the peat was replaced with compost, or 100% compost. The pots were watered and fertilized as required. Analyses consisted of weight of leaves, plant height, stem weight, and total weight exclusive of roots.

Results did not indicate any disease or microbial activity adversely affecting the tobacco plants. Growth results indicated some increases with compost only compared to the other mixtures.

Summary of Miscellaneous Potting Experiments at Fletcher, N. C.

These experiments were initiated in the fall of 1971 to measure the response of selected vegetables and flowers to municipal compost used in potting media mixtures. Information was obtained on germination effects, vegetative growth, and moisture holding capacity. A preliminary germination test with cucumbers, lettuce, and tomatoes indicated no adverse effects from the compost. Experiments were conducted on pot mums; tulips; vegetables including bibb lettuce, tomatoes, and cucumbers.

In regards to growth results on the pot mums, responses to the compost were varied from less shoot growth and bad count to a higher dry weight for the variety Red Anne. In later experiments, in which fertilizer was added, the compost treated pots produced the highest number of shoots and soil only produced the most shoot growth.

With regard to tulips, trends in the growth results indicated that compost produced mixed responses as it did with the mums.

In the experiments with the tomatoes, bibb lettuce, and Gemini cukes, compost did provide beneficial results. Results of these experiments indicated the possibility of compost providing some longer term benefits to the soils as higher 2nd crop yields were noted with the compost treatments as compared to the others. Additionally, compost was shown to aid water holding capacity throughout the experiments. Table 17 provides the chemical analysis of foliage and fruit samples of tomatoes produced in a greenhouse soil mix and in municipal compost.

Selected Demonstrations

Throughout the Johnson City composting project, considerable interest was expressed in "demonstrating" the usefulness of composed mixtures of MSW and sewage sludge. Therefore, a large number of demonstrations were established to show that compost could be used on crops, lawns, strip mines, roadbanks, and other land areas. Results of some of these demonstrations have been reported.

The following figures are photographs depicting selected demonstrations.

Figures 2 and 3 are photographs of 2 of more than 25 demonstrations of burley

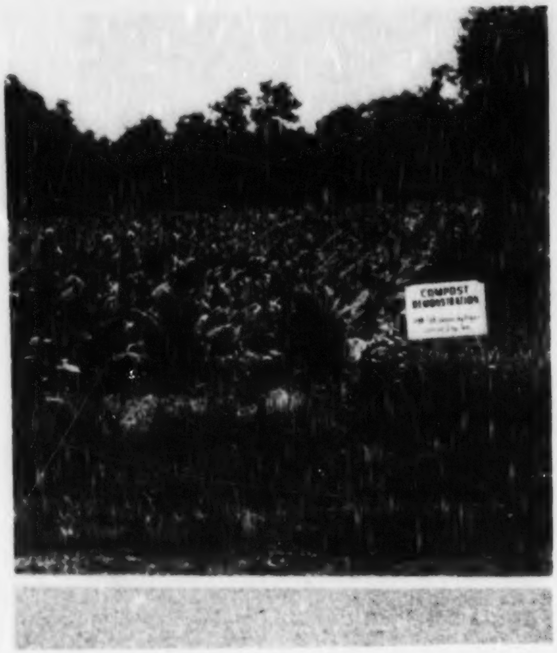

Figure 2. Burley Tobacco Grown in Compost Amended Soil.

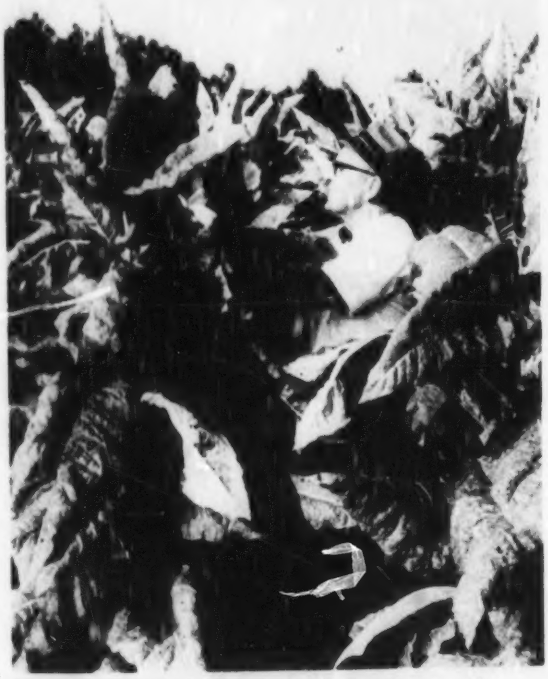

Figure 3. Excellent Growth Response of Burley Tobacco to Compost.

tobacco grown on compost treated soils. Application rates of 12 to 60 tons of compost per acre were used. The results generally indicated that the compost was beneficial and improved crop yield. Since burley tobacco is a relatively high value cash crop, farmers in eastern Tennessee and western North Carolina indicated that they would be willing to pay for compost as an organic amendment. A dollar per ton value was not estimated.

The two center rows of string beans in Figure 4 were treated with compost. While yield information was not available, the photograph shows the treated beans had a definite positive response to the compost when compared to the outer rows.

Bibb lettuce was also grown in compost (Figure 5), although measured responses to the compost did not show conclusively that the compost provided beneficial responses. However, tomatoes responded positively to the compost by ripening quicker and yielding more fruit per vine (Figure 6). Melons were also successfully grown in compost treated soils (Figure 7).

The corn experiment at the Johnson City, Tennessee location was described earlier in the paper. Figures 8 and 9 are photographs depicting the corn growth. As previously reported the corn responded well to the compost with increased yields.

There were numerous individuals who used the better grade compost (i.e., screened compost) as an aid to growing flowers. Figures 10, 11, and 12 illustrate this use. In Figure 10, mums were used in an experiment. Compost in this experiment produced mixed results, but generally compost had beneficial effects on most flowers with the exception of some acid-loving flowers such as Rhododendrons. Compost was used as a mulch in demonstrations on trees, shrubs, and flowers. In these additional demonstrations, compost gave excellent results, especially with roses.

The use of compost to reclaim spoiled land areas resulted in some of the most striking responses. A number of road banks were reclaimed using compost after conventional methods had either failed or gave poor results. Figure 13 is one example. The left side is the area that received the compost. Figure 14 is a photograph showing the results from using compost in routine

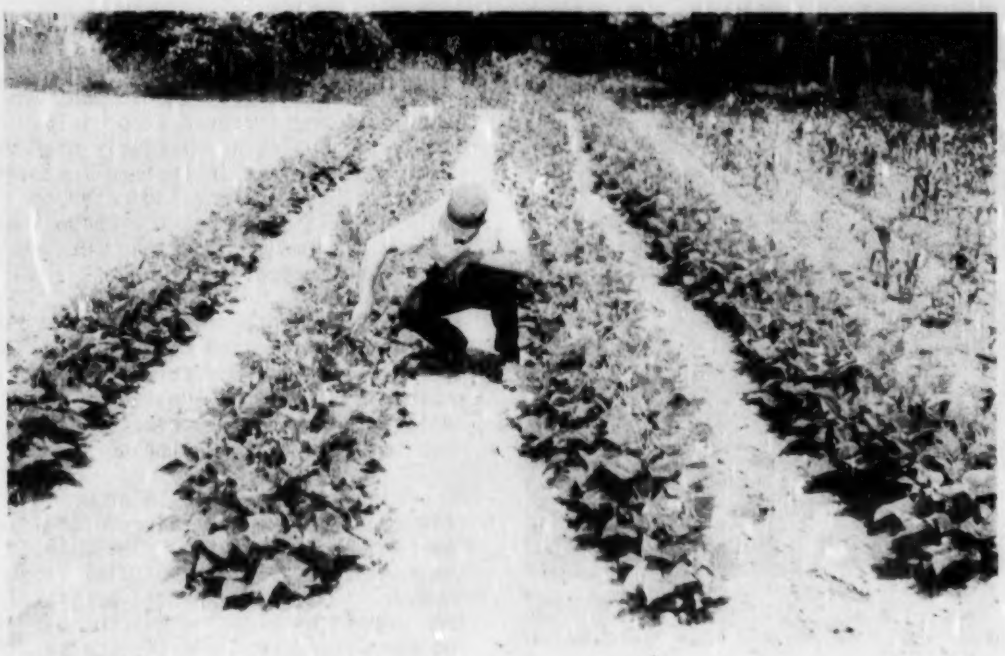

Figure 4. Compost was Beneficial to String Beans.

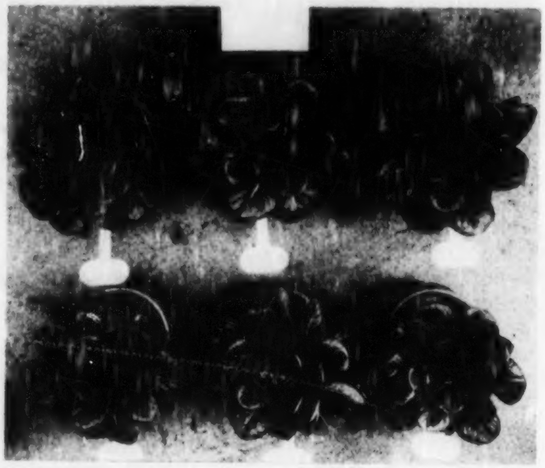

Figure 5. Compost and Bibb Lettuce.

Figure 6. Tomatoes Grew Well in Compost Amended Soils.

Figure 7. Melons also Grew Well in Compost.

COMPOST
DEMONSTRATION

Figure 8. The Corn and Compost Demonstration at Johnson City.

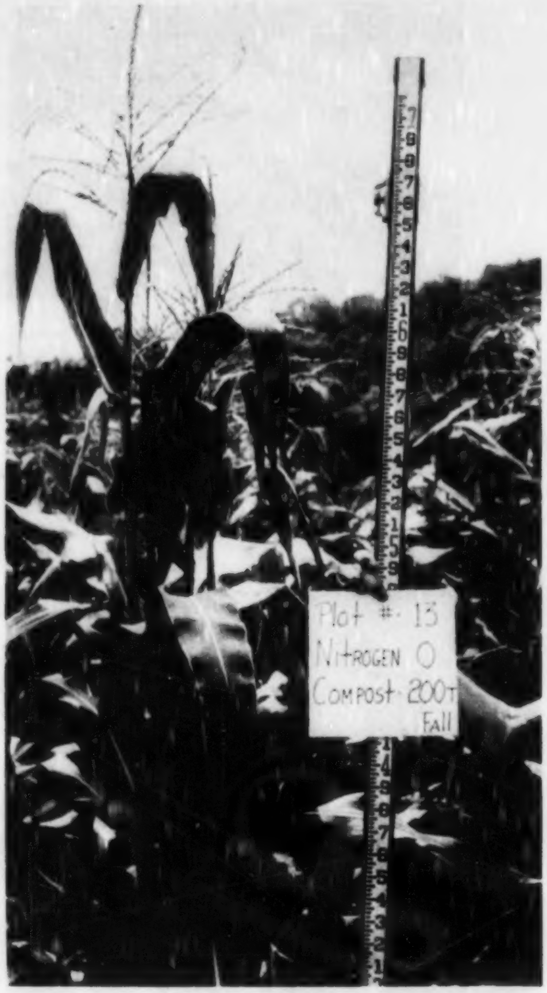
Plot # 13
Nitrogen 0
Compost 200T
Fall

Figure 9. 200 Tons/Acre of Compost Produced Excellent Corn Yields.

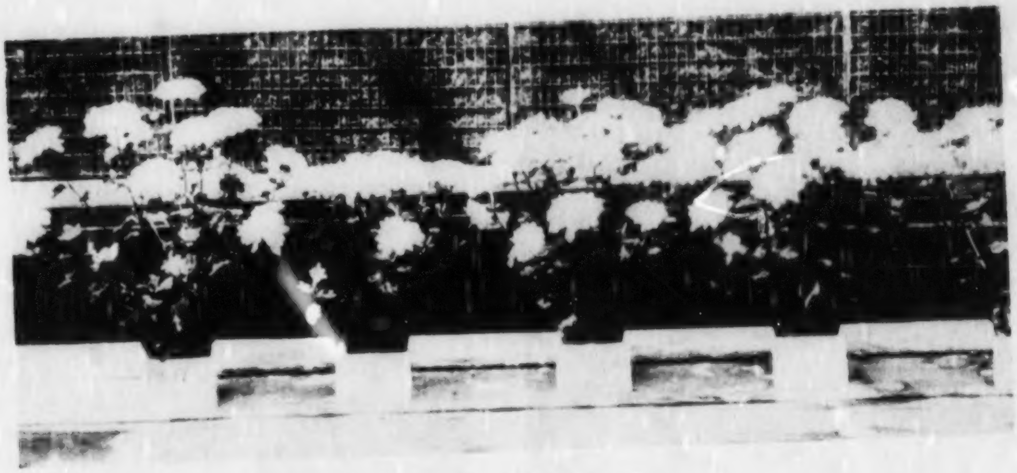

Figure 10. Compost Produced Mixed Results with Mums.

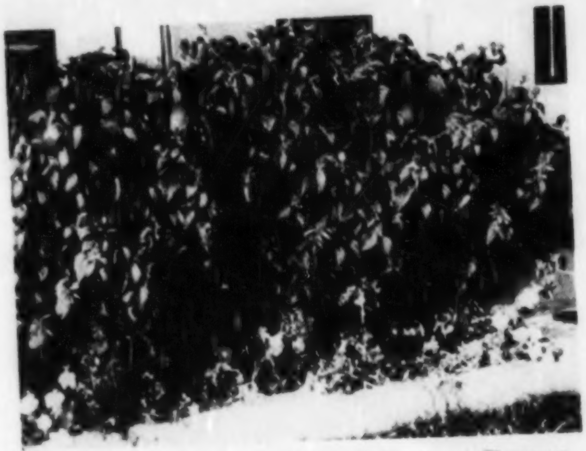

Figure 11. Compost was Used to Grow FLowers at the Compost Plant in Johnson City, Tennessee

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Figure 12. A Beautiful Orchid Grew Well in Compost.

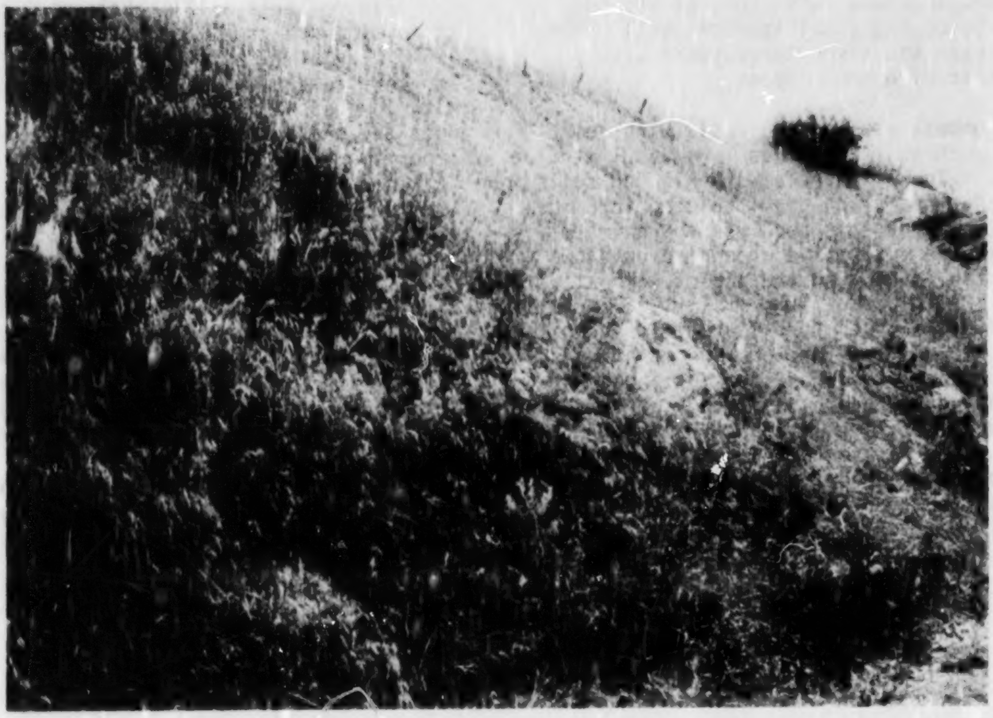

Figure 13. Compost Often Proved Extremely Useful to Reclaim Roadbanks. The Area on the Left was Treated with Compost.

lawn establishment, while Figure 15 shows results obtained with compost after normal procedures had failed. The playground area in the photograph had required additional top soil and reseeding for eight years because of erosion caused by drainage.

Figure 14. Compost was Useful in Establishing Lawns.

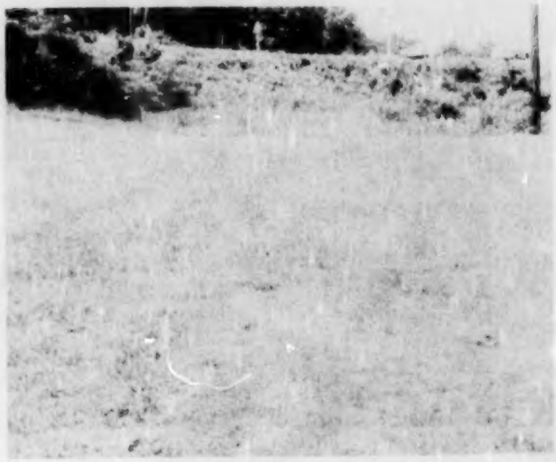

Figure 15. Compost Stabilized this Playground after Eight Years Reported Treatment with Top Soil Failed.

However, treatment with compost stopped the erosion and stabilized the soil. The photograph was taken three years after treatment with the compost.

Compost was good in reclaiming acid soils such as strip mines. However, it also proved helpful in reclaiming high pH soils such as the fly ash pond depicted in Figure 16. The pH of this pond was approximately 11 when the vegetative ground cover shown was established with the aid of compost.

The use of compost to reclaim strip mines yielded the most pronounced effects. Also, because the conditions of those neglected lands were so poor, they were selected as the first demonstration site in 1968. This was so that the unscreened compost could be used without fear of adverse effects from poor esthetics.

Figure 17 is a 1971 photograph of a strip mine area reclaimed with compost in 1968. The still barren ground is apparent in the foreground, while the area reclaimed with compost is also apparent in the back. Note the sharp contrast where the compost stopped.

In addition to the small trees and grass seeds planted in the demonstraion, natural vegetation was established as a result of the compost. This natural vegetation could not become established on the extreme soil conditions depicted in the foreground of the photograph. In 1970, an experiment was initiated to help determine the minimum amounts of compost that could be used to reclaim some of these areas. Figures 18 and 19 are before and after photographs. The photograph shows the visual results obtained from using 26 tons of compost per acre.

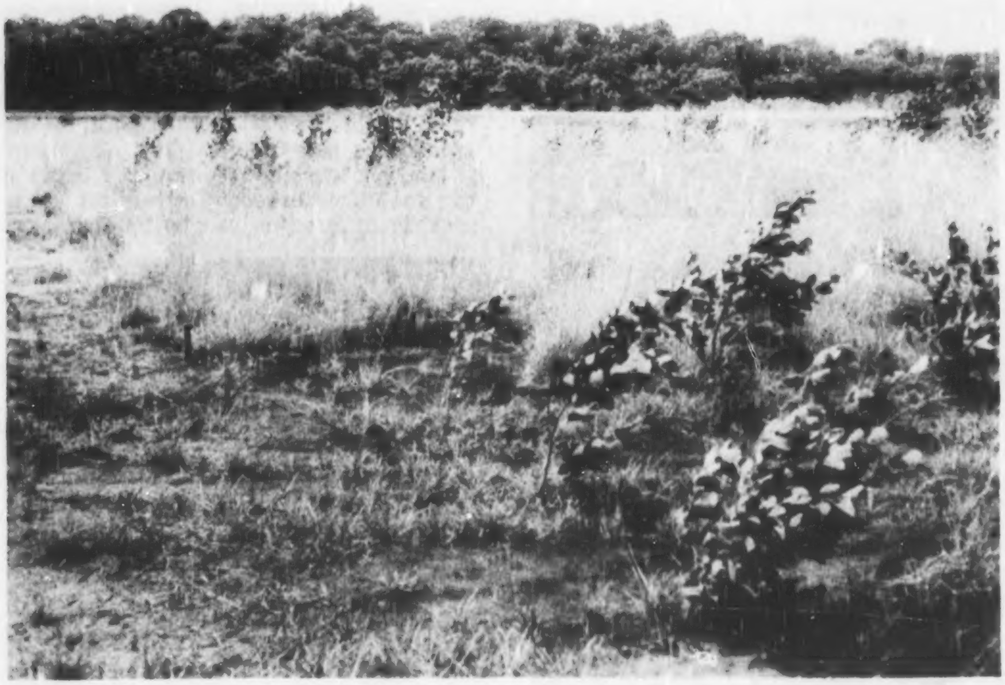

Figure 16. Compost Aided the Reclamation of a Fly Ash Pond with High pH.

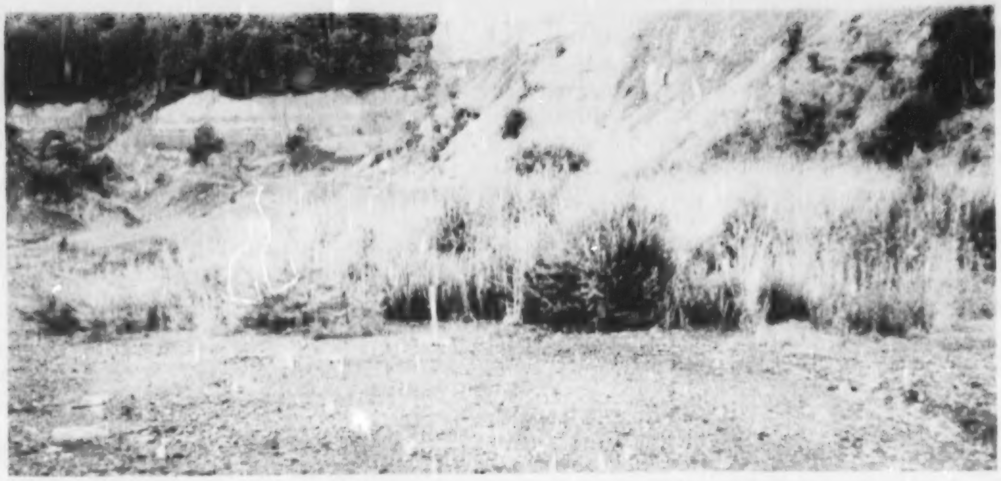

Figure 17. Compost was Exceptional in Reclaiming this Strip-mine Area.

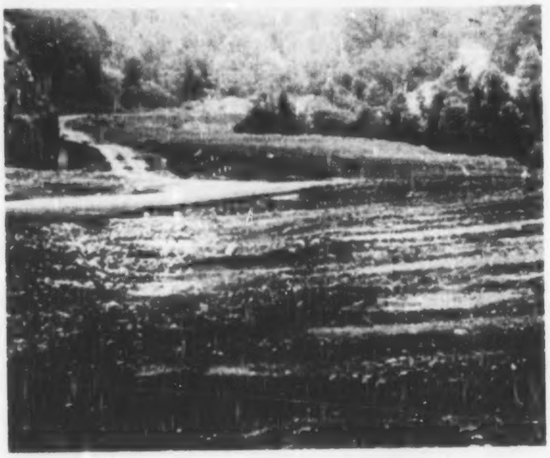

Figure 18. Site of the 1970 Strip-Mine Reclamation Experiment. (Before Reclamation)

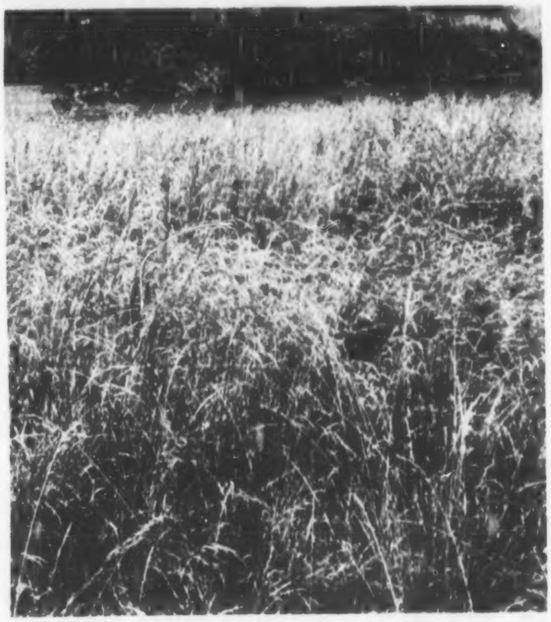

Figure 19. Twenty-Seven Tons of Compost Per Area Produced These Results One Year after Reclamating the Strip-Mine Area in Figure 18.

In general, the many demonstrations conducted showed the versatility of the compost and its value as a soil amendment. Unfortunately, a dollar value for the compost was difficult to establish, as the intangible value may have been more than the real economic value which could be measured from yield responses.

Selected Conclusions

The following conclusions are summarized from the various research projects and "demonstrations" mentioned in this paper or referenced in the bibliography.

Based upon results of the Muscle Shoals experiments, large amounts of municipal compost can be applied in grasslands or croplands with occasional positive yields responses. With respect to the Johnson City corn experiments and others, compost did provide significant positive crop and soil responses indicated by increased crop yields and improved soil characteristics.

Muscle Shoals experiments indicated that application of rather high rates of sewage sludge (>50 metric tons/ha) may increase vegetable production of some species, but this apparent benefit may be accompanied by increased levels of one or more heavy metals. Since certain vegetable species are sensitive to heavy metal accumulation, toxicity may cause decreased yields and high levels of heavy metals in the plants. Although concentrations are generally higher in vegetative tissue, in some instances the reproductive plant parts may be sinks for certain metals. The significance of Cd and Ni concentration, ranging from two to five times those in vegetation from unamended plots, is yet to be determined with regard to food chain implications.

Total heavy metal content of a sewage sludge does not necessarily reflect plant availability. Transformations may occur in soils which increase or decrease availability in these waste products. The organic matter content and composition of the waste material likely influence availability depending upon the stability of the metal complexes formed and the subsequent resistance to decomposition. Apparently the pH value of the sludge per se is not a reliable indicator of plant availability. However, under very acid soil conditions (>pH 5.0), sewage sludge high in Zn content may be toxic to plants. Inorganic sources of Zn initially are more toxic than organic wastes at equivalent rates of application and constant soil pH. The consequence of multiple applications of sludge is unknown. Although heavy metals accumulate with repeated treatment, the organic matrix seems to be protective.

Results show that movement of heavy metals in soil is greater from inorganic than from complexed sources found in sewage sludge under severe leaching conditions; other studies indicate little difference in mobility with normal rainfall and minimal supplemental irrigation.

Although moderate rates of "low risk" sludge appear to pose little hazard to crops, continued use of municipal wastes may ultimately result in heavy metal overloading, especially in some light-textured soils.

Composted mixtures of MSW and sewage sludge were useful in reclaiming soils adversely affected by strip mining, soil erosions, and other factors which caused conditions adverse to plant growth. In this regards, compost increased soil organic matter content, increased soil moisture holding capacity, provided stability against soil erosion, and improved other conditions which might prevent plant growth. Positive responses to compost were documented in a number of varied demonstrations.

RECOMMENDATIONS AND COMMENTS

Recommendations

Municipal waste materials should be carefully characterized with respect to total elemental composition and solubility prior to land disposal. Reactions of organic wastes in soils need further study. Although results suggest that most heavy metals contained in wastes are rather immobile in soils, long-term studies must be forthcoming before the consequence of repeated applications of heavy rates is completely understood.

Although a number of crop species were identified as excluders or accumulators of certain heavy metals, further study is necessary. Routine leaf analysis is not sufficient, since certain edible plant parts other than leaves may be sinks for particular heavy metals.

Based upon the Muscle Shoals research reported, more greenhouse and laboratory work is needed to correlate heavy metal uptake with soil extraction procedures. Existing extractants used for micronutrients may not be suitable for all heavy metals.

There has been little evidence in these studies of heavy metal conversion to more available forms. Organic matter content should be monitored for several more years from plots receiving both single and multiple applications of municpal wastes. Furthermore, soil samples should be extracted during this period to determine whether metal availability to plants is changing. The effect of soil acidification and heating on solubilization and mobilization of metals is being investigated.

Guidelines must be established regarding tolerable levels of heavy metals in food chain components before the significance of elevated levels in crops can be evaluated. It also must be recognized that surface contamination is a contributing factor.

Comments

Based upon responses obtained from applying composted wastes on severely spoiled lands such as strip mines, it would appear that such waste products have tremendous potential for aiding the reclamation of spoiled land areas--even to the extent of placing them back into some useful service.

With respect to the increased crop yields and improved soil characteristics reported in some of these experiments, the results showed that significant positive responses could be obtained from composted wastes. As would be expected, the poorer the soil the larger the response. Also, research and demonstrations at the Johnson City site and other locations indicated that positive benefits could be expected over a prolonged period from single heavy applications of composted MSW. However, with respect to croplands, there still remains the lingering question "what are the total effects to the food chain and ground wastes from heavy metals and other toxic components which might be in the wastes?" The studies summarized here were not of sufficient duration nor scope to answer such questions.

At the time of the Johnson City Composting project, economics and uncertainties would have prevented farmers paying for routine applications of compost to crops. Although, some results did indicate potential longer term economic benefits, the value of these was difficult to demonstrate when compared to the quicker economic returns from commercial fertilizers. There may be situations now where this would be different. Studies being conducted by Dr. Chaney, USDA, investigators in TVA, EPA, and many others to determine and define the complex mechanisms involved in heavy metal migration, transformation, uptake into plants, and associated factors remain extremely important from at least two points:

- If one considers applying composted wastes, including sludges, to improve the crop production and/or reclaim spoiled lands and or
- If one considers applying the wastes to lands as a disposal method.

ACKNOWLEDGEMENT

The work reported in this summary was conducted by different investigators from Tennessee Valley Authority; at Muscle Shoals, Alabama; Johnson City, Tennessee; Mountain Horticultural Research Station, North Carolina; USDA Tobacco Experiment Station, Greenville, Tennessee; and others.

Many organizations and persons were involved in the varied demonstrations conducted. In some cases, tables of results, descriptions, and other information used in this paper were directly excerpted from reports submitted to EPA by these investigators. Their contributions are acknowledged. Additionally, I wish to thank Mr. Robert E. Landreth for providing a verbal presentation in my last moment absence at the conference.

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TABLE 6. CONCENTRATIONS OF SEVERAL HEAVY METALS IN SANGO SILT LOAM (1972)

Treatment	Zn rate, ppm	Soil pH	Concentration in soil, ppm							
			Zn	Cu	Fe	Mn	Cr	Pb	Ni	Cd
Check	0	4.9	4	3	154	100	0.2	10	1.1	0.5
$ZnSO_4$	40	4.9	28	3	153	111	0.3	7	1.3	0.4
	80	4.9	51	2	161	103	0.4	7	1.9	0.5
	160	4.9	106	3	141	90	0.2	11	1.3	0.8
Compost	40	5.7	32	9	247	147	0.9	23	1.3	0.6
	80	5.7	47	15	285	191	1.5	41	2.1	1.0
	160	6.3	80	18	360	203	3.0	49	2.1	1.7
Sludge	40	5.3	29	10	258	222	1.2	28	1.8	2.5
	80	5.3	53	19	363	213	2.0	57	1.4	2.8
	160	5.6	93	27	513	229	5.3	79	2.1	2.9

TABLE 7. DRY FORAGE YIELDS AND CONCENTRATIONS OF SEVERAL HEAVY METALS IN CORN FORAGE AND GRAIN (1972)

Treatment	Zn rate, kg/ha	Dry forage yield, kg/ha	Concentration in corn forage (F) and grain (G), ppm							
			Zn		Pb		Ni		Cd	
			F	G	F	G	F	G	F	G
Check	0	3051a*	41	37	3.4	0.9	4.4	5.1	1.0	0.3
$ZnSO_4$	90	3493a	227	49	3.6	0.7	3.6	6.9	0.9	0.2
	180	4363b	314	57	1.8	0.7	3.1	6.9	0.7	0.2
	360	3230a	475	68	2.9	0.8	3.0	4.1	1.0	0.3
Compost	90	4958bc	77	45	2.3	0.4	3.8	4.1	2.3	0.7
	180	5503cd	98	43	2.0	0.8	2.4	4.6	3.3	0.9
	360	6039d	100	69	3.4	2.7	4.6	2.6	5.3	1.1
Sludge	90	5839cd	94	43	1.5	0.5	2.6	4.0	3.7	0.9
	180	5963d	95	49	1.1	0.6	4.5	4.5	3.5	1.0
	360	5945d	97	44	2.5	1.0	3.1	3.1	4.1	1.2

*Yields followed by the same letter are not significantly different according to Duncan's Multiple Range Test (5%).

TABLE 8. DRY YIELDS AND CONCENTRATIONS OF SEVERAL HEAVY METALS IN BEAN VINES AND PODS (1972)

Treatment	Zn rate, kg/ha	Dry yield, kg/ha		Concentration in bean vines (V) and pods (P), ppm							
				Zn		Pb		Ni		Cd	
		V	P	V	P	V	P	V	P	V	P
Check	0	990a*	1327f	60	45	5.1	1.4	5.0	6.0	0.5	0.2
$ZnSO_4$	90	1252bc	1111f	184	68	3.3	1.2	4.6	6.1	0.5	0.2
	180	1427c	272b	328	79	4.8	1.2	4.6	6.0	0.6	0.2
	360	355a	31a	499	108	3.9	1.3	4.1	5.2	0.6	0.1
Compost	90	1463c	1060e	56	56	4.9	1.4	4.3	4.1	0.5	0.2
	180	1625e	1084e	84	50	4.8	1.2	4.8	4.1	0.5	0.2
	360	1425c	1425f	52	44	4.5	1.2	3.7	3.9	0.5	0.1
Sludge	90	1231bc	1092e	158	61	4.4	0.9	6.7	6.9	1.1	0.2
	180	1234bc	858d	189	75	4.1	1.4	5.5	6.0	1.2	0.2
	360	1537d	597c	164	83	5.5	1.8	5.8	6.9	1.2	0.3

*Yields of either vines or pods followed by the same letter are not significantly different according to Duncan's Multiple Range test (5%).

TABLE 9. CONCENTRATIONS OF SEVERAL HEAVY METALS IN SANGO SILT LOAM (1975)

Treatment	Zn rate, ppm		Soil pH		Concentration in soil, ppm									
					Zn		Cu		Cd		Pb		Ni	
	A*	B	A	B	A	B	A	B	A	B	A	B	A	B
Check	0	0	6.3	5.8	2	2	9	1	0.1	0.1	2	2	0.1	0.1
$ZnSO_4$	40	160	6.0	6.0	7	27	2	1	0.1	0.1	2	2	0.1	0.1
	80	320	5.8	6.0	12	74	1	1	0.1	0.1	2	2	0.1	0.2
	160	640	6.0	6.0	22	129	1	1	0.1	0.1	2	2	0.2	0.1
Compost	40	160	5.5	6.8	6	19	2	7	0.2	0.4	5	10	0.2	0.3
	80	320	6.6	7.0	16	27	6	10	0.4	0.7	8	15	0.3	0.5
	160	640	6.0	7.3	19	63	7	17	0.5	1.2	11	24	0.3	0.6
Sludge	40	160	5.5	5.0	9	69	4	11	0.3	0.7	4	11	0.3	0.6
	80	320	5.7	5.3	19	77	7	15	0.5	0.9	7	13	0.4	0.7
	160	640	6.1	5.7	25	157	110	33	0.8	1.9	11	29	0.5	1.5

*A = 1 application in 1971; B = 4 applications (1971, 1972, 1973, 1974).

TABLE 10. DRY FORAGE YIELDS AND CONCENTRATIONS OF SEVERAL HEAVY METALS IN CORN FORAGE AND GRAIN (1975)

Treatment	Zn rate, kg/ha		Dry yield, kg/ha		Concentration in corn forage and grain, ppm							
					Zn		Cd		Pb		Ni	
	A*	B	A	B	A	B	A	B	A	B	A	B
				Forage								
Check	0	0	7233bc[†]	8147cd	30	46	0.7	0.7	4.2	5.1	0.8	0.8
ZnSO$_4$	90	360	7984cd	7208bc	69	225	0.8	0.8	4.6	5.2	0.8	0.8
	180	720	7047bc	6670ab	93	366	0.6	0.9	4.1	4.5	0.9	1.0
	360	1440	8710d	6637ab	94	600	1.0	0.9	4.2	4.5	0.9	0.8
Compost	90	360	6518a	6418a	38	89	1.9	2.0	5.4	5.0	1.0	0.8
	180	720	6828b	6239a	81	114	1.9	2.5	4.9	5.5	0.8	0.8
	360	1440	7674c	6858b	74	150	2.0	3.5	4.7	5.3	1.1	1.1
Sludge	90	360	6414a	6987b	130	313	3.4	4.9	4.1	5.5	0.9	0.8
	180	720	7038bc	6716ab	158	450	4.7	6.6	5.2	5.1	1.0	0.9
	360	1440	7883cd	7448c	172	400	5.9	7.0	5.7	5.0	1.2	1.1
				Grain								
Check	0	0			36	32	0.2	0.2	0.7	1.1	0.6	0.6
ZnSO$_4$	90	360			40	54	0.2	0.2	1.0	1.2	0.5	0.6
	180	720			49	67	0.3	0.3	1.0	1.1	0.7	0.8
	360	1440			50	68	0.2	0.3	1.1	1.1	0.5	0.6
Compost	90	360			46	46	0.4	0.4	1.6	1.2	0.8	0.8
	180	720			43	46	0.4	0.7	1.0	1.2	0.6	0.8
	360	1440			44	46	0.4	0.6	1.3	0.9	0.7	0.7
Sludge	90	360			45	62	0.8	0.9	1.1	1.3	0.7	0.9
	180	720			48	74	0.9	1.0	1.2	1.1	1.0	0.9
	360	1440			51	64	1.0	1.2	1.0	1.3	1.0	1.1

*A = 1 application in 1971; B = 4 applications (1971, 1972, 1973, 1974).

[†] Yields followed by the same letter are not significantly different according to Duncan's Multiple Range Test (5%).

TABLE 11. DRY YIELDS AND CONCENTRATIONS OF SEVERAL HEAVY METALS IN BEAN VINES AND PODS (1975)

Treatment	Zn rate, kg/ha		Dry yield, kg/ha		Concentration in bean vines and pods, ppm									
					Zn		Cu		Cd		Pb		Ni	
	A*	B	A	B	A	B	A	B	A	B	A	B	A	B
						Vines								
Check	0	0	1658cd[†]	1606c	42	36	9.3	8.7	0.2	0.2	5.4	5.9	2.2	2.2
ZnSO$_4$	90	360	1435bc	1271ab	47	101	[illegible]	9.1	0.2	0.2	5.5	5.6	2.4	2.3
	180	720	1492bc	1232ab	67	226	9.2	[illegible]	0.2	0.3	5.9	6.7	2.1	2.3
	360	1440	1365b	929a	118	288	9.3	[illegible]	0.3	0.4	5.6	5.9	2.3	2.2
Compost	90	360	1538c	1630cd	57	43	9.0	9.3	0.4	0.2	6.0	6.0	2.1	2.1
	180	720	1746d	1685cd	51	46	9.7	9.8	0.2	0.3	5.4	6.1	2.1	2.2
	360	1440	1680cd	1584c	53	50	10.2	9.6	0.3	0.2	6.4	6.5	2.4	2.1
Sludge	90	360	1224ab	1173ab	128	304	10.8	10.4	0.8	1.1	6.7	7.2	4.5	4.4
	180	720	1417b	1028a	141	325	10.6	9.7	1.0	1.4	6.6	7.2	4.9	4.6
	360	1440	1550c	1080a	128	308	10.8	11.0	0.8	1.3	6.7	7.3	3.7	5.5
						Pods								
Check	0	0	558cd	516bc	53	52	10.6	10.5	0.1	0.1	2.4	2.5	2.9	2.5
ZnSO$_4$	90	360	483b	545c	56	70	11.0	11.1	0.1	0.1	2.3	2.7	2.5	2.4
	180	720	524c	549c	61	98	11.0	11.1	0.1	0.1	2.3	2.5	2.3	2.4
	360	1440	508bc	358a	79	107	11.3	11.3	0.1	0.1	2.2	2.5	2.3	2.2
Compost	90	360	610d	582cd	58	60	11.3	12.4	0.1	0.1	2.3	2.3	2.2	2.1
	100	720	534c	544c	56	56	10.5	10.8	0.1	0.1	2.5	2.6	1.8	2.3
	360	1440	529c	548c	54	52	9.8	9.4	0.1	0.1	2.5	2.5	2.2	2.0
Sludge	90	360	402b	429ab	71	111	11.1	11.7	0.2	0.2	2.4	2.4	8.7	7.2
	180	720	429b	326a	82	127	12.3	11.5	0.2	0.3	3.0	2.5	6.7	8.1
	360	1440	438ab	295a	79	115	11.5	12.9	0.2	0.3	2.6	2.7	6.9	8.9

*A = 1 application in 1971; B = 4 applications (1971, 1972, 1973, 1974).

[†] Yields followed by the same letter are not significantly different according to Duncan's Multiple Range Test (5%).

TABLE 12. HEAVY METALS EXTRACTED BY WATER AND ACIDS FROM TUSCUMBIA AND DECATUR SEWAGE SLUDGES

Sewage Sludge	Extractant	Solubility in extractant, % of total				
		Cu	Zn	Cd	Ni	Pb
Tuscumbia	H_2O	0.4	0.6	0.6	0.9	0.1
	0.5 N HNO_3	83	98	76	49	70
	0.5 N HCl	88	94	31	56	69
	DTPA	46	27	35	25	9
Total, ppm*		516	3640	35	43	1560
Decatur	H_2O	1.0	0.7	0.4	0.8	0.1
	0.5 N HNO_3	85	100	100	58	77
	0.5 N HCl	77	100	100	58	73
	DTPA	30	26	45	17	17
Total, ppm*		740	1840	49	40	525

*Determined by dry ashing 2.0 grams of dry sludge for 6 hours at 470 C.

TABLE 13. TOTAL YIELD AND CONCENTRATIONS OF HEAVY METALS IN VEGETABLES, AS AFFECTED BY SLUDGE TREATMENT (1974)

Species	Treatment	Yield kg/plot	Fruit or root conc. ppm Zn	Cu	Cd	Ni	Leaf concentration ppm Zn	Cu	Cd	Ni
Beans	None	7	21	7.9	0.04	1.3	40	6.0	0.46	2.6
(*Phaseolus*	Decatur	7	31	7.6	0.23	2.7	179	7.8	1.70	3.5
limensis)	Tuscumbia	6	28	7.4	0.07	1.8	95	5.8	0.55	2.4
Okra	None	12	40	9.2	0.13	0.7	41	6.9	0.59	1.9
(*Hibiscus*	Decatur	11	61	9.0	0.60	0.7	94	10.0	2.00	2.8
esculentus)	Tuscumbia	11	43	8.3	0.16	0.7	55	7.1	0.59	1.8
Peppers	None	23	23	10.4	0.09	1.0	68	19.0	0.71	1.7
(*Capsicum*	Decatur	25	41	13.4	0.40	2.3	143	21.0	2.70	2.7
spp.)	Tuscumbia	20	31	10.5	0.14	1.6	92	18.0	0.76	2.1
Tomato	None	23	15	3.4	0.12	1.8	49	19.0	0.66	1.5
(*Lycopersicon*	Decatur	29	24	3.8	0.39	3.3	77	22.0	2.10	2.3
esculentum Mill.	Tuscumbia	30	20	2.9	0.20	0.7	55	18.0	0.75	1.6
Squash	None	43	47	12.8	0.03	0.9	93	14.0	0.34	1.7
(*Cucurbita*	Decatur	67	83	15.3	0.20	2.6	233	19.0	0.63	4.0
pepo L.)	Tuscumbia	73	93	12.8	0.15	1.6	226	15.0	0.36	2.2
Turnip	None	13	39	5.5	0.42	1.8	52	6.3	0.59	2.9
(*Brassica*	Decatur	14	133	8.9	1.30	2.8	194	9.4	2.60	4.0
rapa)	Tuscumbia	13	73	6.5	0.42	2.1	96	7.3	0.59	2.9
Radish	None	--	48	3.2	0.29	3.0	56	5.5	0.92	3.9
(*Raphanus*	Decatur	--	149	3.8	0.92	3.7	275	7.5	3.10	6.0
sativus L.)	Tuscumbia	--	121	3.3	0.33	2.6	271	3.8	0.88	5.1
Kale	None	11	-	--	--	-	33	6.0	0.63	1.8
(*Brassica*	Decatur	11	-	--	--	-	161	8.1	2.30	3.5
oleracea)	Tuscumbia	13	-	--	--	-	113	7.6	0.63	2.4
Lettuce	None	--	-	--	--	-	71	12.9	1.00	2.2
(*Lactuca*	Decatur	--	-	--	--	-	212	22.8	8.60	3.8
sativa L.)	Tuscumbia	--	-	--	--	-	336	21.0	3.00	3.9
Spinach	None	--	-	--	--	-	168	9.1	1.00	2.3
(*Spinacia*	Decatur	--	-	--	--	-	163	12.7	2.80	2.9
oleracea)	Tuscumbia	--	-	--	--	-	225	10.0	0.84	2.1

TABLE 14. TOTAL YIELD AND CONCENTRATIONS OF HEAVY METALS IN VEGETABLES, AS AFFECTED BY SLUDGE TREATMENT (1975)

Species	Treatment	Yield, kg/plot	Fruit or root conc. ppm Zn	Cu	Cd	Ni	Leaf concentration ppm Zn	Cu	Cd	Ni
Okra	None	13	48	10.7	0.42	1.4	56	15.4	0.[illegible]7	1.8
	Decatur	16	55	10.7	1.20	1.4	63	14.1	3.[illegible]0	1.7
	Tuscumbia	13	49	10.7	0.38	1.4	76	14.1	0.44	1.8
Pepper	None	8	32	12.1	0.04	1.5	61	14.7	1.04	2.3
	Decatur	8	43	10.7	0.60	2.4	153	15.4	2.92	3.0
	Tuscumbia	6	40	10.1	0.12	1.3	130	15.4	0.78	2.4
Tomato	None	7	34	11.4	0.33	1.1	38	21.4	1.70	2.5
	Decatur	12	37	11.4	1.12	1.2	79	19.4	6.70	2.8
	Tuscumbia	9	36	11.4	0.40	1.3	61	22.8	1.70	2.2
Squash	None	38	65	12.7	0.27	1.5	112	16.1	0.70	2.6
	Decatur	40	90	12.7	0.72	1.9	233	12.1	2.15	4.4
	Tuscumbia	30	88	12.7	0.19	1.7	287	9.4	0.87	2.9
Lettuce	None	4	--	--	--	-	109	17.4	1.20	3.9
	Decatur	5	--	--	--	-	166	18.1	7.00	4.4
	Tuscumbia	4	--	--	--	-	316	12.7	2.60	4.9

TABLE 15. ANNUAL TREATMENTS OF COMPOST AND NITROGEN, 1969-1973

Treatment[a]	Annual Application	Season Applied	Weight Per Subplot (lb) Compost	Nitrogen
T-1	-0-	--	--	--
T-2	80 lb, N	Spring	--	2.2
T-3	4 tons, compost	Spring	90	--
T-4	160 lb, N	Spring	--	4.4
T-5	8 tons, compost	Spring	180	--
T-6	8 tons, compost	Fall	180	--
T-7	4 tons, compost	Fall	90	--
		Spring	90	--
T-8	8 tons, compost + 80 lb, N	Spring	180	2.2
T-9	8 tons, compost + 160 lb, N	Spring	180	4.4
T-10	8 tons, compost	Fall	180	--
		Spring	180	--
T-11	50 tons, compost	Fall	1,125	--
T-12	100 tons, compost	Fall	2,250	--
T-13	200 tons, compost	Fall	4,500	--

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a. Phosphorus (P_2O_5) and potassium (K_2O) supplements and 200 pounds of 0-26-26 were added to each subplot each year.

b. Nitrogen - ammonium nitrate sulfate, 30-0-0-5S (N-P-K-S).

TABLE 16. ELEMENTAL COMPOSITION OF COMPOST USED IN THE JOHNSON CITY CORN EXPERIMENT

Element	Concentration
Nitrogen (N)	13,000 ppm (1.30%)
Phosphorus (P)	2,600 ppm (0.26%)
Potassium (K)	9,700 ppm (0.97%)
Carbon (C)	273,000 ppm (27.30%)
Calcium (Ca)	46,000 ppm (4.60%)
Magnesium (Mg)	6,000 ppm (0.60%)
Sodium (Na)	4,595 ppm
Copper (Cu)	312 ppm
Manganese (Mn)	449 ppm
Zinc (Zn)	1,073 ppm
Moisture Content	40%

TABLE 17. CONCENTRATIONS OF ELEMENTS IN FOLIAGE AND FRUIT PORTIONS OF GREENHOUSE TOMATOES PRODUCED IN A GREENHOUSE SOLID MIXTURE AND COMPOST.* (MHCRS - Fletcher, N.C. - 1972)

Element	Greenhouse Soil Mixture		Municipal Compost	
	Foliage	Fruit	Foliage	Fruit
	PPM	PPM	PPM	PPM
Iron	449	128	153	113
Aluminum	106	5.8	57	11.4
Copper	14.9	7.6	27	12.5
Manganese	392	18	187	17
Nickel	5.3	6.9	4.8	7.0
Zinc	96	30	143	38
Mercury	0.02	ND	0.05	ND
Lead	6.0	32	4.8	31
Arsenic	0.5	ND	ND	ND
Cadmium	2.6	0.3	5.6	0.5
Chromium	3.7	3.7	5.2	3.6
Selenium	0.05	0.02	0.04	0.02
Vanadium	ND	ND	ND	ND
Boron	103	17	92	21

*Oven-dry basis

**ND - not detected

LAND CULTIVATION OF MUNICIPAL SOLID WASTE

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INTRODUCTION

Soil is a natural environment for the deactivation and degradation of many waste materials; and all soils exhibit a capacity to deactivate and degrade wastes, though not at the same rate. Land cultivation of waste material is a disposal technique by which wastes are mixed with the surface soil to promote aerobic decomposition of the waste's organic content. The technique is also known as landspreading, refuse farming, and soil incorporation.

This paper presents a progress report on an Environmental Protection Agency (EPA)-sponsored study of land cultivation being performed by SCS Engineers (EPA contract No. 68-03-2435). Discussed will be background information, experiences with land cultivation of municipal solid waste, including a case study, and associated environmental effects.

BACKGROUND

Waste deposition on land continues to be the most widely-used disposal alternative for most types of residues. Land disposal is expected to further increase as regulations prohibiting discharge of sewage sludge to the ocean are implemented, increased pretreatment of industrial wastewaters is required, and more stringent air pollution control technology is imposed.

Land disposal for most solid wastes and sludges presently involves burying the residues in landfills. While sanitary landfills have been and continue to be economically attractive, they are not without actual and potential environmental and social problems.

For example, as the papers in this symposium amply demonstrate, the anaerobic degradation of organic materials that occurs in buried wastes usually results in a variety of largely unpredictable and unquantifiable by-products: gas and leachate. If uncontrolled, these by-products can adversely affect groundwater quality and the surrounding environment. Also, waste degradation and settlement continue for many years in a sanitary landfill, necessitating continued long-term maintenance of the site. This maintenance can be costly and often raises difficult legal problems. In addition, once the site has been used as a sanitary landfill, its future uses are limited to continued waste disposal or a small number of post-landfill development options. These shortcomings can usually be overcome, but the remedy is generally measured in terms of dollars.

In light of these real and potential shortcomings associated with the conventional sanitary landfill disposal method, it is not surprising that alternative disposal techniques have been proposed and implemented. Land cultivation is one such alternative.

Land cultivation has been looked at as a means of aerobically decomposing organic waste to reduce its volume and to prevent the formation of unwanted gases and to minimize the intensity of leachate problems. Also, the processes could be carried out repeatedly on the surface of a disposal site, thereby "recycling" the land. Under the most ideal conditions, proponents of land cultivation claim that the site could be returned to any other land use including agriculture after cessation of disposal activities.

Although ideal conditions are rarely met in the field, and literature on the environmental impacts, regulatory controls, and waste types and characteristics in relation to land cultivation is scarce, the practice of land cultivation has enough promise and has had enough preliminary successes that many industries and municipalities have started to land cultivate their wastes or are planning to do so.

EPA project 68-03-2435 deals with both municipal solid wastes and industrial wastewaters and sludges. For the purposes of this symposium, however, land cultivation of municipal solid wastes will be of primary concern. In addition to the information gathered from a literature review and from interviews with knowledgeable professionals, results from a case study are also included.

LAND CULTIVATION PROCEDURES

Land cultivation has been practiced most extensively by the petroleum refinery industry. Oily wastes (including API separator sludges and tank bottoms and drilling muds) are routinely disposed of by land cultivation at several refineries. Also, operators of at least two commercial industrial waste disposal facilities are now land cultivating various types of wastes, primarily hydrocarbons.

The process of land cultivation appears to work in a wide range of climatic conditions. However, warm, humid climates offer the most favorable conditions since biodegradation of the organic fraction is enhanced with adequate moisture and high temperatures. Land cultivation has been used in cold and dry climates, but the waste degradation rates are relatively slower.

Land cultivation procedures at any location are basically the same:

- A suitable site is located and prepared.
- Waste is deposited at one end of the site and spread by track dozer in thin (10 to 20 cm) layers over the surface.
- Waste is physically mixed with the soil by use of a farm plow, disc, or rototiller.

SITE FEATURES

Land cultivation sites are relatively flat, with slopes generally less than five percent. Berms or dikes are provided to prevent surface runoff from sites receiving industrial sludges and wastewaters.

The soils at these sites vary over a wide range of texture and drainage characteristics. One ongoing site, for example, started operation in beach sands, although by now the drilling muds have significantly changed the texture of the surface soil.

Land preparation generally entails scarification of the surface to expose as much soil area as practical. Vegetation is usually removed, but smaller bush and grass may be left in place to be mixed with the waste. Grasses in the disposal plot will become established if the plot is left idle for some time.

WASTE SPREADING AND MIXING

There are two basic objectives of the land cultivation process:

- Increase the availability of the waste for microorganisms, and
- Aerate the soil-waste mass to ensure that sufficient oxygen is available for biodegradation.

As a result, waste decomposition will proceed at a rapid rate, and hazards associated with leachate and gas generation can be minimized.

Mixing methods and frequencies vary depending on site specific conditions: waste types, climate, soil characteristics, presence of nutrients, and moisture content. In some cases, it may be unnecessary to cultivate the waste with soil at all after initial mixing. At existing sites, mixing intervals vary from once per week over several weeks to twice per year. In general, the site operator judges from the site's visual appearance when the waste-soil mix should be recultivated. At some sites, samples of the mixture are analyzed to aid in this determination.

Conventional farm cultivation equipment is generally adequate to accomplish the incorporation of industrial wastewaters and sludges into the soil. However, mixing of municipal solid waste with the soil requires a heavy-duty soil stabilizer such as those used in road construction.

LAND CULTIVATION OF MUNICIPAL SOLID WASTES

In comparison to sanitary landfilling, very little information is available on land cultivation of municipal solid waste. However, limited data are available on the land application of shredded raw waste and refuse compost.

Municipal solid wastes that have been experimentally land cultivated are composed primarily of paper, metals, glass, and organics such as food wastes (garbage) and yard trimmings (Table 1). Of course, each locality has its own characteristic mix of wastes depending on local conditions.

An early research study by Hart, et al. (6) incorporated coarsely ground, unsorted municipal refuse into surface soil at Davis, California, at rates of 112 to 896 metric tons/ha (50 to 400 tons/ac) dry weight. Nitrogen fertilizer was added to balance the carbon-to-nitrogen (C/N) ratio of the refuse, and the plots were kept moist. After one year, an unidentifiable organic residue plus identifiable fragments of glass, metal, and plastic remained. No odor, insect, or rodent problems were reported, but some blowing of paper and plastic occurred. The second year, it was somewhat difficult to incorporate an additional 896 metric tons/ha of waste material into the soil since the surface layer primarily consisted of residue from the previous year's waste application. The cost for this land cultivation activity was estimated to be from \$2.77 to \$6.20/metric ton (\$2.51 to \$5.63/ton).

A land cultivation program in Oregon indicated that municipal solid waste can be most easily handled if it has first been shredded or pulverized (13). This study found that the shredded waste should be distributed evenly over the land surface at rates such that it can be readily incorporated into the soil. Blowing debris can be controlled by use of overhead sprinkler irrigation if immediate incorporation is not possible (14). It was noted that with conventional field tillage equipment, an application rate of 448 metric tons/ha (200 tons/ac) should not be exceeded (1, 5). With application of 896 metric tons/ha, the unconsolidated refuse was approximately 60 cm thick. After mechanical compaction and irrigation, the refuse layer was reduced to a thickness of 20 cm.

TABLE 1. COMPOSITION OF MUNICIPAL SOLID WASTE, U.S. AVERAGE (12)

Material Categories	10^6 tons	10^6 metric tons	Composition, percent of total
Paper	53.0	48.1	36.8
Glass	13.5	12.2	9.3
Metals	12.7	11.5	8.8
Ferrous	11.2	10.2	7.8
Aluminum	1.0	0.9	0.7
Other nonferrous metals	0.4	0.4	0.3
Plastics	5.0	4.5	3.4
Rubber	2.8	2.5	1.9
Leather	1.0	0.9	0.7
Textiles	1.9	1.7	1.3
Wood	4.9	4.4	3.4
Subtotal nonfood products	94.8	86.0	65.6
Food waste	22.4	20.3	15.6
Yard waste	25.0	22.7	17.4
Misc. inorganic waste	1.9	1.7	1.3
Total	144.0	131.0	100.0

Very little of the sandy soil near the Boardman, Oregon, site was mixed with the refuse using the Howard rotavator when the refuse depth exceeded 15 cm. During cultivation, rags wound themselves around the rotavator shaft (8, 18). All the studies at Oregon indicated that the application rates for shredded municipal waste depend on waste composition and plans for final land use.

In another study, King et al. (8) applied unsorted, shredded municipal refuse along with anaerobically digested sewage sludge to a Guelph loam soil in Ontario, Canada, at rates of 188 and 376 metric tons (207 and 414 tons) and 2.3 and 4.6 cm (0.9 and 1.8 in), respectively, per ha. The refuse was first spread rapidly on the soil surface; a furrow was plowed about 30 cm deep into which most of the refuse adjacent to the furrow was raked by hand. The next furrow was then plowed to cover the refuse. This technique resulted in good refuse coverage, but concentrated a relatively large amount of the refuse at the 15- to 30-cm depth. Refuse in the 0- to 10-cm layer was well mixed with the soil by subsequent discings, but there was little mixing of refuse at the lower depths. Following this refuse application, sewage sludge was applied, allowed to dry, and then disced into the soil to a depth of 10 cm. Although it was not possible to physically mix the sludge with the refuse to achieve a favorable C/N ratio, the application technique used did place the lower C/N material in an area of high root uptake and the high C/N material at the lower level where nitrate moving downward was reduced due to immobilization and/or denitrification.

Stanford (10), currently under contract with EPA (OSWMP), has initiated a three-year study near Houston, Texas, on a multivariate trial to assess the effects over time on crop yield and quality, soil quality, and water quality of adding shredded municipal refuse, dry sewage sludge, and chemical fertilizer separately and together. Shredded municipal refuse (80 percent less than 20 cm nominal size) and dry sludge were applied at rates up to 560 metric tons/ha (250 tons/ac) and 336 metric tons/ha (150 tons/ac), respectively, to a sandy clay (pH 5.3). The wastes were incorporated into the soil by rototilling with a heavy-duty soil stabilizer which is equipped with mixing blades. Clover and grasses were then seeded. Initial observations showed marked differences in growth due to waste application; high application rates produced only sparse vegetation (Stanford, personal communication).

In the Tri Service Project at the Navy's facilities in Pt. Hueneme, California, (Durlak, personal communication), paper waste consisting mostly of cardboard was shredded to three different sizes (0.6-3.81, 10.2-15.2, and 31 cm) and applied at rates of from 44.8 to 448 metric tons/ha (20 to 200 tons/ac) to two soils (sandy and clayey). The waste was incorporated into the surface 0 to 46 cm (0 to 18 in) deep by a soil stabilizer. Usually one pass was sufficient to adequately mix the refuse with soil. Researchers on this project concluded that land cultivation is not cost-effective ($15/ton or $16.53/metric ton), in comparison with other disposal methods.

SOIL INCORPORATION OF COMPOSTED MUNICIPAL SOLID WASTE

In a sense, land cultivation of municipal solid waste is akin to composting the refuse in thin layers on the land surface. Information concerning soil incorporation of compost is also being investigated. Most of this data concerns revegetation potential; no specific information on environmental effects, land cultivation techniques, or costs are available.

Several investigations have evaluated land application of compost to reclaim drastically disturbed land and enhance plant growth.

The composting process reduces the C/N ratio of the refuse, stabilizes the organic materials, and eliminates many of the health hazards associated with raw refuse. If the compost includes sewage sludge, it usually contains small but significant amounts of nitrogen and phosphorus, which serve as nutrients for soil microorganisms and plants. On the other hand, composting may concentrate certain trace elements to hazardous levels (14).

Refuse compost applied at 35 and 70 metric tons/ha (16 and 31 tons/ac) added organic matter and plant nutrients to sand tailings from phosphate mining at Bartow, Florida, as shown by the subsequent growth of sorghum and oat crops on the treated tailings (7).

A number of studies have used the refuse compost from Johnson City, Tennessee, to reclaim strip mine spoils in Virginia (3, 9), an alkaline abandoned ash pond (4), and an acid eroded copper basin soil material (11). Revegetation was possible in all these trials.

CASE STUDY - SOIL ENRICHMENT PROGRAM

The city of Odessa is located in a semi-arid area, approximately 516 km (315 mi) west of Dallas, Texas. Like many other cities, Odessa is faced with the situation where the existing sanitary landfill is nearing capacity. Available new sites near the city are small, while larger sites are located at considerable distance from the city. To dispose of the 191 metric tons (210 tons)/day of solid wastes generated, the city has undertaken a land cultivation program. This program has a dual objective; i.e., disposing of wastes and at the same time improving the soil fertility by increasing soil organic matter content and water holding capacity of soil-waste mixture. It is hoped that this waste disposal by land cultivation will restore the grazing capacity of the rangeland.

Currently, the city is leasing two sites, comprising a total of 607 ha (1,500 ac) west of Odessa. About 92 percent of the city's refuse is shredded to <20.3 cm (8 in) at a site within the city, and 26 percent (about 50 metric tons/day) is trucked 8 km (5 mi) to the land cultivation field. Odessa's goal for 1977 is to cultivate 75 percent of the refuse, ultimately rising to 90 percent.

Initial waste applications were made in the summer of 1974 on 12.5 ha (31 ac) of land at the rate of 100 metric tons/ha (40 tons/ac). These initial applications were mixed into the soil by tandem disc behind a standard farm tractor. In October 1975 a field trial was conducted to select a rototiller for use at the site. From these trials, a heavy-duty Buffalo Bomag MPH-1 soil stabilizer was chosen for use (Figure 1). Wastes were spread with the front spreading blade of the soil stabilizer, then incorporated into the loamy soil by the rototiller in the rear. A second pass is made later to more thoroughly mix the soil and shredded refuse.

From December 1975 to December 1976 wastes were applied to a total of 28 ha (70 ac) at rates ranging from 88 to 312 metric tons/ha (35 to 124 tons/ac). Current refuse application rates average 126 metric tons/ha (50 tons/ac); the city is still working to determine the optimum rate.

In addition to refuse, Odessa is also applying sewage sludge and septic effluent on various plots at the land cultivation field alone or in combination with refuse. The cost was estimated at $6.03/metric ton ($5.47/ton) for land cultivation of municipal refuse as compared to $8.85/metric ton ($8.03/ton) for landfill.

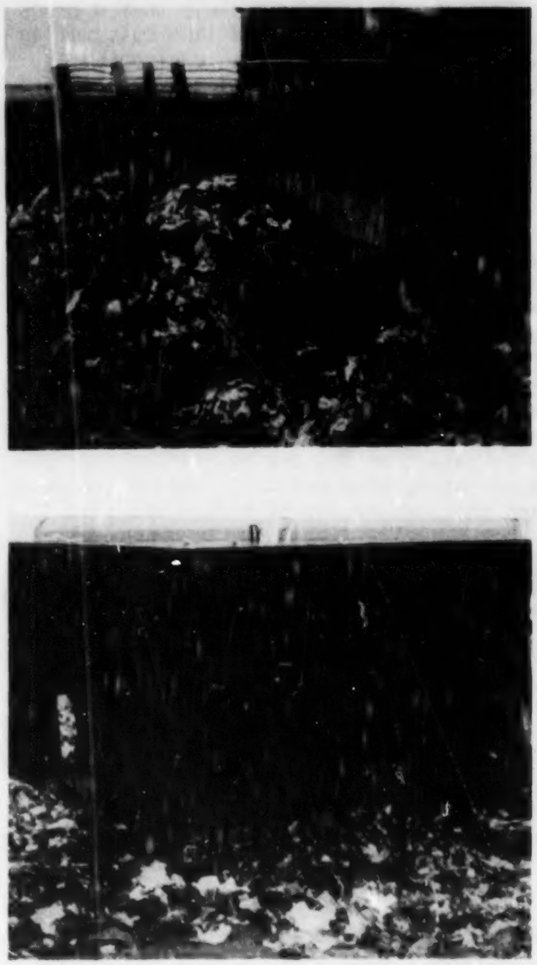

Figure 1. Municipal refuse is being spread and mixed by a soil stabilizer (Buffalo Bomag MPH-1)

Wastes are cultivated year round, with only one application on each plot of land. After soil incorporation, five ranch grasses are sown on the plot to establish grazing for cattle. It is planned to return the land to grazing when the grasses become established.

PROBLEMS ASSOCIATED WITH LAND CULTIVATION OF MUNICIPAL SOLID WASTES

Since there is little experience with land cultivation, many of the associated problems can only be surmised from experimental work and the one ongoing project in Odessa, Texas. In fact, some proponents of this disposal technique claim that a properly managed and situated land cultivation site would pose less of an environmental threat than a conventional sanitary landfill. Aerobic decomposition of refuse, which contains relatively low concentrations of heavy metals, nutrients, and toxic substances, would tend to minimize occurences of water or air pollution instances, advocates feel. For example, in both field and column leaching studies, Halverson (5) found that movement of heavy metals and phosphorus was limited, and with proper management practices, the nitrate movement from the refuse-treated soil could be controlled and should not present a problem to groundwater contamination. (Proper management practices include compatible site selection and design, refuse application rates, wind and soil erosion controls, and thorough mixing of refuse and soil.)

Problems associated with land cultivation, both observed and expected, include the following:

EQUIPMENT BREAKDOWNS

Mechanical mixing of the wastes with soil by a soil stabilizer often results in rags and wires being wrapped around the blades of the rototiller (Schnatterly, personal communication). The severity of this problem may be minimized if the refuse is shredded to the dimensions (<5 cm) recommended by Volk.

AESTHETIC PROBLEMS

Blowing of paper, plastic films, and other light materials has been reported. Also, the land cultivation site itself may appear like an open dump (especially using coarsely ground refuse) at least until the waste's organic fraction is degraded. Odors are not a problem although refuse odors may be apparent before aerobic decomposition begins. Rodent attraction and fly propagation are not reported.

ENVIRONMENTAL CONTAMINATION

In humid areas, the relatively large uncovered refuse area exposed to direct precipitation may become over-saturated and begin to leach. Both the raw and the decomposed refuse can contribute contaminants that may move to groundwater. However, available information shows that leachate from aerobically decomposed refuse contains generally less contaminants than leachate generated in a landfill (5). Also, air emissions from a land cultivation site should be no greater than those from a sanitary landfill receiving the same waste.

"HARDWARE DISEASE"

If the land cultivation field is to return to pasture, as in Odessa, there is potential health hazard to grazing animals resulting from ingestion of metallic, rubber, and other waste materials. Volk (personal communication) suggests that large metallic objects be sorted out and the wastes shredded to 5 cm or less in dimension in an effort to solve this "hardware disease" problem.

EFFECTS ON PLANT GROWTH AND UPTAKE OF ELEMENTS

As noted, one reason given for land cultivation is the potential for returning the land to other uses, particularly agriculture. Thus, it is important to determine the extent to which vegetation grown on land cultivated plots are affected by the presence of refuse and its degradation by-products.

As part of this study, surface soil and vegetative samples were taken from control and refuse-treated plots at Odessa, Texas, in January 1977 for chemical analysis. Its objective was to determine the effects of land cultivation of municipal solid waste on soil chemical properties and elemental uptake. The soil occurring in the land cultivation field is primarily a calcareous, well-drained Ratliff loam. There are a variety of native weeds and grasses in the area, but due to the limited scope of the field study, only two species, wild geranium (_Erodium cirutarium_) and wild buckwheat (*Eriogonum annum*) were sampled.

Results from soil analyses showed that land cultivation of shredded municipal refuse increased soluble salt (EC) content and HCl-extractable manganese and zinc appreciably in the soil (Table 2). Halverson (5) reported the concentrations of a number of elements, including sodium, iron, manganese, zinc, copper, boron, phosphorus, and organic nitrogen in a Sagehill loamy sand increased with refuse additions (up to 896 metric tons/ha). The concentrations of all the constituent analyzed for the soil samples taken from the Odessa site are in the low range of typical soils and should not pose phytotoxicity or water pollution problems.

TABLE 2. CHEMICAL CHARACTERISTICS OF SURFACE SOILS FROM THE CONTROL AND REFUSE-TREATED PLOTS AT ODESSA, TEXAS

Constituent[1]	Control	Treated
pH	7.65	7.63
EC, mmhos/cm	0.61	2.12
TKN, %	0.043	0.066
Org. C, %	0.41	0.69
	- - ppm - -	
P	22.5	25
Na	110	122
B	<0.2	0.25
Mn	22.5	33.1
Mo	0.05	0.05
Zn	1.8	7.2
Pb	0.09	0.12

[1] Electrical Conductivity (EC) and B were measured in the saturation extracts; elements in ppm were determined in 0.1N HCl extracts.

Plant analysis data indicate that wild geranium and buckwheat contained adequate nitrogen with and without refuse treatment (Table 3). Except for boron in wild geranium, there were no significant differences in plant uptake due to land cultivation. In a more detailed study, Cottrell (1) reported that with addition of 896 metric tons/ha shredded municipal refuse to a Sagehill sand, the uptake of zinc and boron by wheat and fescue approached or exceeded phytotoxic levels during the first growing season. He also found that molybdenum uptake by alfalfa reached levels potentially hazardous to livestock during the first growing season, but decreased to normal in the second year.

Based on these preliminary findings, it is concluded that the soil chemical properties were not significantly affected by the application of shredded municipal refuse at Odessa and that concentrations of the trace elements analyzed for wild geranium and buckwheat grown at the disposal site are within the ranges generally reported for grasses.

SUMMARY AND CONCLUSIONS

Land cultivation of municipal solid waste has received very little attention, probably due to the lack of data on the economics, productive uses, and associated environmental problems. Optimum application rate is about 224 metric tons/ha (100 tons/ac). Land application of other municipal solid wastes such as lime and alum sludges have not been reported in the literature. Based on its chemical properties, lime sludge could probably be used in place of limestone as a liming material on acid soil.

Incorporation of refuse compost into barren lands stabilizes soil structure and enriches organic matter content making revegetation possible, particularly with addition of chemical fertilizers. Generally, however, equivalent yield increases on productive agricultural land can be obtained more economically with inorganic fertilizers than with largely unproven methods such as the use of solid wastes or refuse compost.

A "soil enrichment program" has been initiated at Odessa, Texas, which incorporates shredded municipal refuse into the surface soil alone and in combination with sewage sludge and septic effluent. Problems encountered include equipment breakdowns during waste incorporation, impaired aesthetics (unsightliness), and possible development of "hardware disease" in grazing cattle. These problems may be solved if the refuse is shredded to a dimension of 5 cm or less and thoroughly incorporated into the soil. The soil chemical properties and forage quality at the land cultivation field in Odessa are not significantly affected by the disposal practice.

TABLE 3. ANALYSES OF WILD GERANIUM AND WILD BUCKWHEAT GROWN ON CONTROL AND REFUSE-TREATED PLOTS AT ODESSA, TEXAS

Element	Wild Geranium		Wild Buckwheat	
	Control	Refuse	Control	Refuse
	- - % - -		- - % - -	
N	3.46	4.27	2.31	2.27
P	0.275	0.388	0.400	0.550
	- - ppm - -		- - ppm - -	
Na	550	525	650	785
B	6	25	15.9	17
Mn	32.1	36.7	25.4	32.5
Mo	2.0	2.0	0.50	0.50
Zn	68.5	67.3	56	62.3
Pb	5.6	5.7	4.2	6.4

Based on available information and field study results on the management and disposal of municipal solid waste, the following conclusions can be drawn:

1. Land cultivation as a disposal alternative is practiced only on a limited scale in the U.S. The trend indicates that there will be no significant increase in the future.

2. Land cultivation is viable only where soil, climatic, and environmental conditions enable the shredded refuse to be left partially covered for extended periods without possibility of environmental contamination. The disposal program should be related to a land reclamation project, as at Odessa, Texas.

3. For optimum results, the municipal solid wastes should be shredded to a relatively small size (5 cm or less in dimension). Presently, this degree of shredding is prohibitively expensive.

4. Since municipal refuse has a relatively high C/N ratio, application of nitrogen (and possibly phosphorus) fertilizers may be necessary to enhance microbial decomposition of the waste, and to supply the nitrogen requirement of plants of revegetation if agricultural production is to be part of the project. To this end, sewage sludge could be applied together with municipal refuse since it contains available nitrogen and phosphorus and free moisture to enhance biodegradation of the waste material.

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IMPLICATIONS OF PRICE INCENTIVES FOR SOLID WASTE MANAGEMENT

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ABSTRACT

The use of market-related incentive (or disincentive) mechanisms has received scant consideration as pollution control policy in the United States, particularly in the area of solid waste management. This paper reviews what is known about incentive mechanisms for controlling air and water pollution and suggests possibilities for pricing (user charges) in the management of municipal solid waste. Empirical estimates of price and income elasticities are stated. It is hypothesized that incremental user charges for solid waste collections and disposal would reduce the waste generation rate, enhance source separation of recyclable materials, accelerate technological innovation, and minimize total system costs. A properly structured price or user charge would be an equitable way of allocating public resources and could be an efficient system for maximizing net social benefits of a municipally provided service. Further research is needed to delineate the conditions under which user charges would be economically feasible for solid waste management.

INTRODUCTION

The relevance of this subject to a Symposium on gas and leachate in sanitary landfills may not be altogether clear, except as the theme of this Symposium suggests a general concern over possibly adverse effects from mismanagement of solid waste. People are concerned about the increasing amounts of solid waste for various reasons: to some it means deteriorating landscapes and aesthetics; others emphasize the potential threats to health through contamination of water supplies; still others think largely in terms of conservation and wasted resources.

The actual adverse effects embodied in these concepts are difficult to quantify in an aggregate sense; thus, the related effects are mostly descriptive and with a tendency toward use of emotional terms. There is, however, an indicator that suggests the importance of solid waste, and that is public outlays or costs. Expenditures for managing solid waste rank seventh among state and local budget items on a per capita basis (1). Overall expenditures for pollution control of solid waste (including collection and disposal) were estimated at over $5 billion in 1975 (2). These large outlays for solid waste management mean that other badly needed community services and projects must be reduced or postponed.

The increasing amounts of solid waste and mounting public costs for managing it frequently evokes the question "What can be done about it?" There is no general concurrence on this issue and no broad public policy has as yet been established to deal with the problem. Implicit in the language of past national legislation was the need for source reduction and conservation, but government research and implementation program efforts to date have concentrated on handling and disposing of solid waste after it has been generated (3). In recent years, the recapturing of materials from the waste stream, including converting combustible materials to energy has received increased attention. These programs, however, continue to neglect the issue of source reduction and may be only "second-best" solutions for achieving resource conservation.

A quick glance at past research activities will show that large sums of public monies have been expended to develop technology for managing solid waste after it has been generated, while there has been a great lack of research on the source of the problem--the solid waste generator. Of some 500 available publications listed by the Office of Solid Waste (U.S. EPA), over 30 percent relate to solid waste collection and disposal. Only about a dozen reports touch on source reduction, and these mostly on technological possibilities for waste reduction. The kind of source reduction referred to in this paper relates to the behavioral changes needed to bring about a reduction in the current post-consumer solid waste generation rate of 3.75 pounds per day, and the projected rate of 5 pounds per day by 1990 (4).

PAST RESEARCH ON ECONOMIC INCENTIVES FOR SOLID WASTE MANAGEMENT

Someone might ask "What more is there to investigate about the behavior of solid waste generators?" After all, we have studied the attitudes of housewives toward solid waste and recycling, the problem of abandoned automobiles, and throw-away beverage containers, to mention a few. What has been generally lacking in these studies, however, is an in-depth analysis of the behavior of waste generators, in a framework of demand for collection and disposal of their solid waste.

It may be useful, before embarking upon a discussion of the implications of incentives, to define what is meant by an "incentive." The dictionary defines an incentive as "that which incites to action." In an economic context, an incentive is that which encourages individual or corporate action in expectation of a potential gain or loss. A price is a form of incentive or disincentive mechanism which the decision-maker translates into potential gains or losses.

A price may relate to either the supply or demand side of a market activity. In the area of solid waste, the U.S. EPA's re-research has been mostly with regard to the supply of collection and disposal services; for example, the use of wage incentives to increase worker productivity (5). The results of this study should be useful to large cities which tend to have municipally-operated collection and disposal services (6).

Studies on the demand for waste collection/disposal services have addressed incentives for the most part in only a tangential fashion; for example, the relationship between backyard vs. curbside pickup, the relationship between frequency of pickup and volume of waste generated, or the effects of educational and publicity campaigns on source separation (7). The feasibility of using prices to modify the attitudes and behavior of solid waste generators has been generally overlooked.

Perhaps the first attempt to study the relationship between charges and waste generation was by Hirsch, using data for the St. Louis County-City area (8). His methodology and approach did not fully address the issue of user charges, however. The cost model included number of pickups as a proxy for quantity; thus average cost of pickup was represented. This variable turned out to be nonsignificant (assumed constant). Thus, the higher cost observed for the user charge probably reflected only the associated administrative costs and not the net effects of the charge system. At the time of the study, only 7 of the 24 communities in the St. Louis area had a "user or service" charge, and none of these were incrementally structured.

The University of Chicago, under a grant from the U.S. Public Health Service, conducted a study on the relationship between income and residential refuse (9). The particular emphasis in that study was on the relationship between income and residential waste quantities set out for collection. Price effects could not be observed as the City of Chicago financed collection/disposal service from general tax revenues.

The study showed that quantity of waste was positively related to income, with an average per capita income elasticity of 0.53 for selected weeks. Thus, a a ten percent increase in real income suggested a five percent increase in household solid waste set out for collection. The elasticity ratio appeared to have a seasonal pattern; however, with a lower ratio in winter months and also lower for low income households. The researchers suggested that residential refuse may consist of two components: (1) a "basic" quantity that is invariable with regard to income and season of the year, and (2) an "excess" component that is more sensitive to "income and race."

A recent U.S. EPA supported study by the University of California investigated the demand for waste collection in 20 California communities (10). The results indicated a price elasticity of demand of 0.44. This suggests that a ten percent increase in price or user charge would reduce waste set out for collection by 4.4 percent. The accuracy of the estimate must be taken with caution, however, because average price was used as a proxy for incremental price. Since average price tends to overstate marginal price as perceived by consumers, the price elasticity ratio was probably underestimated.

The California study had several deficiencies including inadequate sample size. The number of communities with an incremental charge (i.e., a charge per bag or can) was limited to four, and only one community (San Francisco) had a charge structure approaching a true user charge. The possibility that simultaneity affected the results must also be considered. Thus, the results based on California communities can hardly be indicative of a national response to user charges.

USE OF INCENTIVES FOR AIR AND WATER POLLUTION CONTROL

Incentives for pollution control have received considerably more attention in Europe than in the United States. The effluent charges assessed by the Genossenschaften (Water Resources Associations) in the Ruhr industrial area of West Germany are frequently cited as an example. These charges are not true user charges in the strict sense, however, as the charges tend to be based on the average cost of treating effluent (11). The French system more nearly meets the requirement that the charge relate incrementally to the benefits received. The effluent charges are based on the impurity of the effluent, and additionally, there are fees for users of water based on the quantity and quality of water withdrawn. Thus, the system also provides information on the consumer's "willingness to pay" for purer water.

In the U.S., user charges specifically directed at water and air pollution control are practically nil. The results of an empirical study on the use of sewer surcharges may have important implications for water pollution control, however (12). Data from 34 cities having sewer surcharges were analyzed by the Water Resources Research Institute of the University of North Carolina. The results showed that: (1) a 45 percent reduction in pounds of BOD could be expected from a modest surcharge of 2.7 cents per pound of BOD where no surcharge existed previously; and (2) an additional increase of 10 percent in the surcharge would result in a further reduction of 8 percent in the BOD loading. This suggested an average price elasticity of about 0.90, starting with a zero charge. The response to sewer surcharges does not necessarily imply that a similar response would result from stream charges; but it does lend credence to the theory that incentives such as effluent charges will be included in production decisions affecting waste generation and disposal (13).

An advantage of incentives that should not be overlooked is the potential stimulus they provide for technological innovation. This kind of stimulus for process changes and waste treatment or control is not provided by a direct regulatory approach. By that approach, the inclination of waste generators is to sit back and let the government suggest the technology to meet environmental standards. This puts the onus on the governmental regulatory authority to be the "know-it-all" technology expert.

Implementation of user or effluent charges for water pollution abatement is just beginning in the U.S. The impetus for this is provided by the Water Pollution Control Act Amendments of 1972 (14). Section 204 of the Act requires that a municipality initiate a user charge reflecting the user's contribution to the total waste load and associated costs in order to qualify for federal assistance in the construction of water treatment facilities.

Our experience with incentives for air pollution control is even less than in the water area; however, an interesting example of the potential for incentives was reported recently (15). The case involved the electric utilities, and the suppliers and distributors of fuel oil for these utilities. The energy crisis and fuel shortage in the winters of 1972-73 and 1973-74 motivated suppliers and distributors in the

New York City area to request variances or exceptions to the existing air pollution laws on the justification that low sulfur fuel oil was in short supply. The New York City EPA perceived that granting the variances could result in windfall profits to suppliers and distributors because it would permit delivery of a lower priced fuel at a previously contracted delivery price. The Agency, however, had to depend on the industry for the information concerning the availability of low-sulfur fuel oil. The true situation became known when the New York City EPA decided to grant variances but with the stipulation that the City receive a surcharge of 75 cents per barrel for fuel oil containing between 0.3 and 1.0 percent sulfur, and $2.00 per barrel for oil containing between one and two percent sulfur. The effect of this was that by the end of the 1972-73 winter one of the major suppliers had found it necessary to deliver only a third of the amount of high sulfur oil it originally estimated was needed, and another major supplier was able to limit his distribution to half the amount originally estimated.

IMPLICATIONS OF A PRICE POLICY FOR SOLID WASTE MANAGEMENT

The American Public Works Association in its recent publication indicated that perhaps 30 percent of the cities in the U.S. rely on some form of service charge to finance solid waste collection (16). The APWA notes that the use of user charges may be declining. The trend is obscured, however, because municipalities define a user or service charge in various ways. A recent study by Columbia University reported that only about 13 percent of the communities surveyed had an incremental or variable user charge (17). The percentage of national population affected was even lower as large communities and cities depended more heavily on general taxation.

Given that a user charge is an equitable method of charging for a service, the reasons why its use is not more prevalent are not entirely clear. Aside from the usual objections to user charges; for example, that solid waste collections benefit the public in general and therefore should be financed by general taxes, or that such charges are regressive, there are probably also more subtle reasons why municipal officials are reluctant to adopt user charges. Possibly, financing public services by general taxation maintains a climate in which members of society can hope that somehow their individual cost is less than the average taxpayer's. Society as a whole must assume the costs in proprotion to the services demanded, but it has been shown that local politics are such that individual costs can be disproprotionate to the benefits obtained (18). Another reason why local officials may resist user charges is because they lack information on the ultimate effects.

Studies by the U.S. EPA suggest that user charges for solid waste management are theoretically feasible (19). Studies by academic researchers generally supported this thesis (20). Moreover, the general conclusion is that public officials by rejecting incremental user charges have encouraged households to generate larger quantities of waste (21). The perceived need for pricing solid waste management is not limited to academia, however, as statements to this effect by persons having close contact with solid waste problems can frequently be found in the press.

RESEARCH NEEDED TO EVALUATE USER CHARGES

The key question for which an answer is needed is "Does a properly structured incremental user charge result in lower total system costs and increased net social benefits when compared with other systems?" Any policy recommendations for or against pricing solid waste must first be able to answer that question.

In the U.S. EPA's Decision-Makers Guide in Solid Waste Management, it is suggested that the basic issues in solid waste management can be categorized as costs, environmental factors, resource conservation, and institutional factors (22). User charges are given as one alternative which may enable localities to balance costs with revenues, make citizens aware of management costs, and provide impetus for efficient operations. On the other hand, the Guide notes that user charges are more complex to administer and can cause "problems" for people on fixed incomes. The feasibility of user charges as a waste reduction policy is not mentioned; however, and local decision-makers are not provided with the set of conditions under which user charges would or would not be efficient.

The following aspects of a user charge

system are important to local solid waste managers in reaching a decision:

1. Will an incremental user charge provide incentive for a reduced waste generation rate; and if so, in what form will the reduction occur and what will be its magnitude?

2. Will an incremental user charge provide the incentive to households to source separate recyclable materials; and if so, what will be the effect on an optimal design for end-of-pipe resource recovery systems?

3. Will an incremental user charge provide incentive to waste managers to adopt efficient management techniques; and if so, will this include increased innovation of new technology?

4. Will an incremental user charge induce waste generators to switch to alternative disposal options, including food disposers, compactors, composting, burning, littering, and increased bulk set-outs; and if so, what costs to society are associated with these alternatives?

5. Will an incremental user charge impact on household purchases of optional levels of service (e.g., set-out, frequency, pickup location, etc.)?

SUMMARY

The research literature suggests that effluent charges and emission fees are feasible incentive instruments for controlling water and air pollution. The potential of user charges as a similar incentive strategy for solid waste management has not been seriously considered nor researched. User charges for collection of residential solid wastes might substantially reduce the quantity of waste having to be disposed of or recycled, with resulting lower total system costs. This might be achieved, for example, through an increased propensity by households to source separate recyclable wastes, increasing the importance of decisions regarding type of packaging and product durability, and of decisions affecting the generation of yard wastes, such as planting and fertilizing of grass, trees and shrubs. A few communities presently employ various pricing schemes, but the full effects of these are not fully known. Policy recommendations concerning user charges should not be made in the absence of this information.

In summary, the key issue is whether implementation of a user charge system for solid waste collection makes a community and the Nation better or worse off than before. Implicit in this is the question of whether individual freedom of choice is preserved. A user charge system retains the concept of consumer sovereignty while regulatory approaches such as mandatory deposits, production or disposal taxes, and licenses or permits based on approved technology and discharge standards reduce individual freedom of choice. A fully structured user charge is generally accepted as an equitable method of allocating public resources.

Widespread adoption of user charges might not only reduce solid waste quantities and adverse effects on the environment, but could also mean an overall reduction in expenditures for solid waste, thus permitting scarce resources to be shifted to other social needs such as education, medical research, and rapid transit. The waste generation rate in San Francisco is cited as an example of what an incremental user charge system can achieve (23). Household solid waste collections in that city amounted to 699 pounds per capita in 1970, in contrast to the average of 937 pounds for California communities where general revenue financing predominated. In San Leandro, California, another community having an incremental user charge, the residential solid waste amounted to 60 percent less than in communities without a user charge (24).

To fully research the effects of user charges will require substantial funds. The parameters of variables comprising the complete system should be determined. Ideally, this would involve field experiments where the variables of interest are controlled. The greater investigative power of such experiments would produce a larger payoff in predictive reliability. This kind of research, directed toward human behavior--the real source of the solid waste problem--is long overdue.

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EFFECTS OF DECOMPOSITION GASES ON LANDFILL REVEGETATION AT TVA'S LAND BETWEEN THE LAKES

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INTRODUCTION

Landfilling is the most common method for disposing of solid refuse in the United States, but information on special methods or techniques required for revegetation of such sites is very limited. On its Land Between The Lakes, the Tennessee Valley Authority (TVA) in 1975 began research on the revegetation of a sanitary landfill to improve the appearance and allow productive reuse of the land for the benefit of people and wildlife. The project involves determining the various kinds of gases that result from decomposition of landfilled solid wastes, measuring the relative volumes of these gases, and assessing their effects on the survival and growth of various species of grasses, shrubs, and trees.

TVA developed Land Between The Lakes in southwestern Kentucky and northwestern Tennessee as a national demonstration of the utilization of a region's available resources for a combination of outdoor recreation, resource management, and environmental education. It is a continuing development and demonstration project, and the new concepts and methods being tried and tested at Land Between The Lakes are helping establish criteria for the development of improved recreational facilities and environmental education programs throughout the United States.

Land Between The Lakes is a 170,000-acre peninsula between Kentucky Lake on the Tennessee River and Lake Barkley on the Cumberland River. Roughly 40 miles long and 8 miles wide, it is about 85 percent forested and has 300 miles of shoreline.

The recreation potential of this site has been partially developed with numerous family campgrounds, lake access areas, and group camps. More than 2.1 million visitors were attracted to Land Between The Lakes in 1975.

About 20 percent of those who visit Land Between The Lakes participate in its formal and informal environmental education programs, for the area also provides an outdoor classroom for enhancing environmental awareness. School groups can come for resident or single-day programs in environmental eduation at the 5,000-acre Environmental Education Center. Casual visitors are also provided opportunities for learning experiences in the management of timber and fish and wildlife, control of flooding and erosion, and development of environmentally desirable technologies such as the use of solar energy.

Because of the relative isolation of the area and because of its size and the large number of visitors it receives, Land Between The Lakes operates and maintains its own water supplies, wastewater treatment facilities, solid waste collection system, and sanitary landfill. In keeping with the emphasis on environmental education, these facilities are used to develop and demonstrate environmentally acceptable principles and practices of sanitary engineering.

PURPOSE OF RESEARCH AND DEMONSTRATION PROJECT

The principal difference between landfills and other disturbed sites is that covered organic refuse produces decomposition gases in varying quantities. These decomposition gases include carbon dioxide, methane, and smaller amounts of hydrogen, carbon monoxide, nitrogen, and oxygen. According to Flower (1975)[1], the survival, growth and condition of plants have been unusually poor at landfills where moderately high concentrations of gas have been detected.

TVA began research at the Land Between The Lakes sanitary landfill in late 1975 to test the adaptability of various species of grasses, trees, and shrubs to conditions associated with sanitary landfills; to obtain information on variability in tolerance of various plants to landfill conditions; and to select species, families, and individuals that demonstrate superior potential for landfill revegetation for use in breeding and for use in revegetating these areas. The project involves identifying the decomposition gases produced in the Land Between The Lakes landfill, measuring their volumes, and evaluating their effects on the survival and growth of vegetation.

DESCRIPTION OF LANDFILL

The present 16-acre landfill has been in continuous operation since its establishment in 1966. At first, area landfilling was the primary method used; however, since 1971, the trench method has been used. During the peak of the recreation season, as much as 70 tons of solid waste is disposed of in the landfill. The waste load at other times is about 2 tons per week. The waste is high in organic content from food residues, fish scraps, and packaging materials. Soil at the landfill site is of the Tuscaloosa Formation (cretaceous) and is composed of unconsolidated clay, silt, sand, and gravel. The thickness of the soil at the landfill site exceeds 75 feet. It is underlain by the Fort Payne Chert (Mississipian).

According to the United States Geological Survey report on the region, water in the overburden of this formation generally has a pH of about 6.5, a hardness of about 75 mg/l, and a bicarbonate content of 75-100 mg/l. Ground water level is estimated to be 50 feet below the surface.

PROCEDURES

A plot 184 feet wide and 312 feet long was selected. Sixteen 20-foot square subplots were located over a recently finished refuse cell that had an average depth of 15 feet.

Varied species of woody plants that can be established on disturbed sites or that have been successful in artificial reforestation were chosen for trial:

Shrubs

Runner oak (*Quercus pumila*, *Q. minima*)
Bear oak (*Q. ilicifolia*)
Dwarf chinkapin oak (*Q. prinoides*)
American beautyberry (*Callicarpa americana*)
Silky dogwood (*Cornus amomum*)
Common chokecherry (*Prunus virginiana*)
Black-fruited chokecherry (*P.V. melanocarpa*)
Allegheny chinkapin (*Castanea pumila*)
Autumn-olive (*Elaeagnus umbellata*)
Western sand cherry (*Prunus besseyi*)

Trees

Sycamore (*Platanus occidentalis*)
Carolina buckthorn (*Rhamnus caroliniana*)
European alder (*Alnus glutinosa*)
Tupliptree (*Liriodendron tulipifera*)
White ash (*Fraxinus americana*)
Persimmon (*Diospyrus virginiana*)
Chinese chestnut (*Castanca mollissima*)
Sawleaf zelkova (*Zelkova serrata*)
Sawtooth oak (*Quercus acutissima*)
Redbud (*Cercis canadensis*)

The experiment with woody plants involved a randomized complete block design with separation of the tree and shrub species within each block. Six replicates were used, four on the landfill and two as controls. On the landfill, 12 plant row plots were used with 8 by 8-foot spacing. The area available for the control blocks was limited, and reductions were made to four plant plots per species with a 6 by 6-foot spacing. All shrubs and trees were planted in March and April 1976.

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In addition to the woody plants, six species of grasses--redtop (*Agrostis alba*), timothy (*Phleum pratense*), annual ryegrass (*Lolium multiflorum*), perennial ryegrass (*L. perenne*), orchardgrass (*Dactylis glomerata*), and tall fescue (*Festuca arundinacea*) var. Ky 31--were tested. The grasses were planted in a randomized complete block design using 20 by 20-foot plots and two replications of the six species. Grasses were sown in September 1975 at a rate of 0.5 pound per plot (equivalent to a rate of 54 pounds per acre). The plots were fertilized at a rate of 400 pounds per acre with 15-15-15 fertilizer and were moderately mulched with oat straw.

EVALUATION

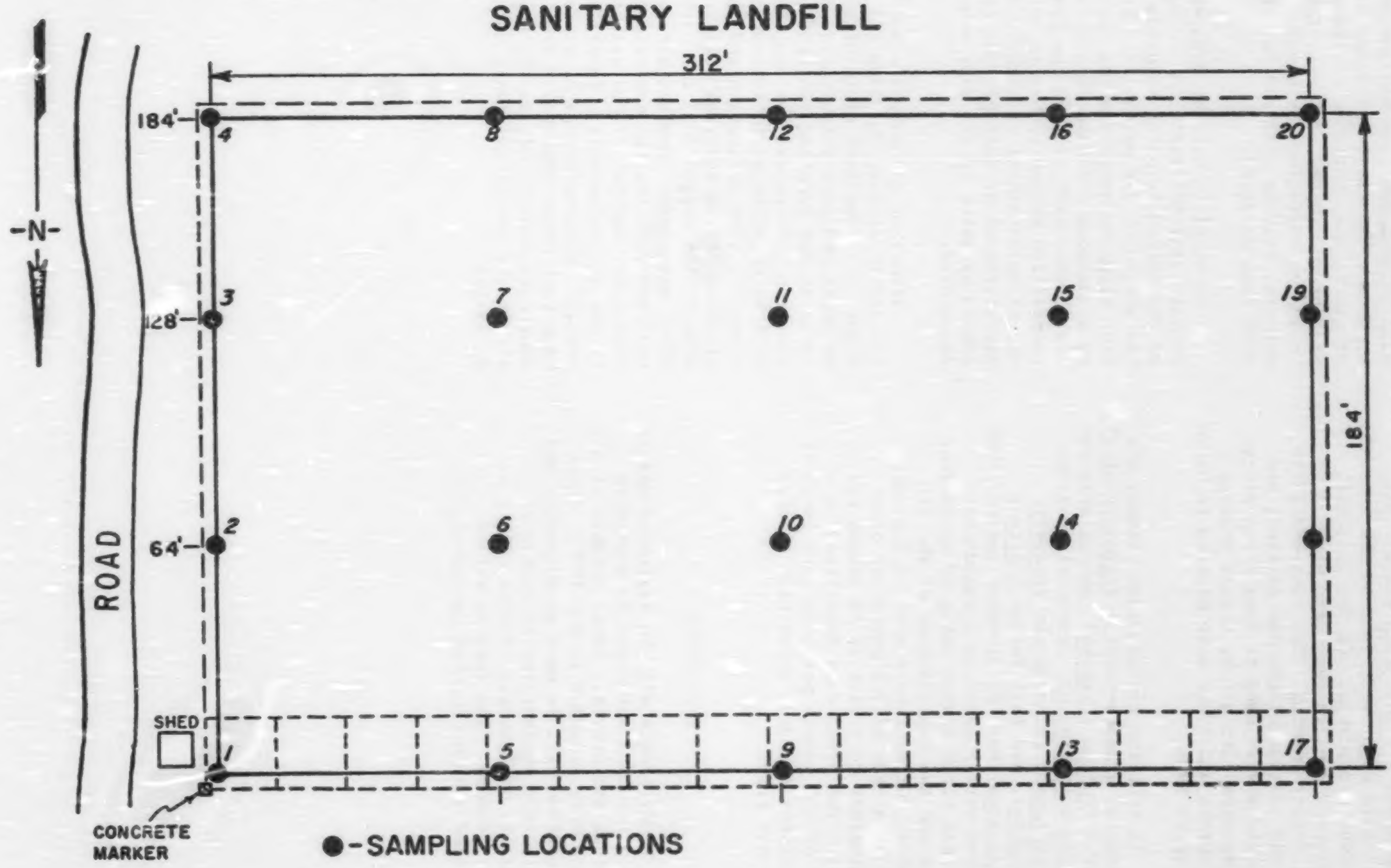
Woody plants will be evaluated yearly on a plant-by-plant basis to determine survival and growth. Total heights of all plants were recorded in May 1976. Other evaluations will be made as necessary and may include evaluations of traits of flowering or fruiting, foliar color, or other factors. The results will be analyzed and interpreted in correlation with information from the gas monitoring program. The composition and concentration of gas in the root zone of the woody plants will be determined, and vegetative response, including variations by species or seed sources, will be analyzed for correlation with data on gases.

Grass plots will be evaluated yearly in late fall for variation in cover density as determined by point sampling techniques. Production of dry matter will be determined from clipping samples as necessary. Data on herbaceous response will be analyzed for correlation with data on the quantity and composition of gas in the root zone, and visual observations of effects such as discoloration of foliage that might be caused by gases in the root zone will be documented.

Temporary gas sampling points were located throughout the landfill (see Figure 1). The test plots of grass will be monitored most intensively to establish criteria for sampling frequency and location. Permanent gas sampling stations will be established to improve the efficiency of data collection and the quality of data. Field analyses will be performed with fyrite oxygen (O_2) and carbon dioxide (CO_2) analyzers. Combustible gas will be analyzed with the MSA explosimeter. Additional samples of gas will be analyzed by gas chromatography. In preliminary results, concentrations of explosive gases have been below detectable limits, and concentrations of CO_2 and O_2 have been within the non-stress range (CO_2 <2%, O_2 18-20%).

FIGURE 1

GAS SAMPLING LOCATIONS AT LAND BETWEEN THE LAKES SANITARY LANDFILL



312'
184'
184'
128'
64'
-N-
ROAD
SHED
CONCRETE MARKER
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
19
20
● -SAMPLING LOCATIONS

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